A

APPENDIX A

Public Participation Materials

Presentation Outline

- Onterra, LLC
- Why create/update a Lake Management Plan?
- Elements of a Lake Management Planning Project
  - Data & Information
  - Planning Process

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Lake Management Planning

Onterra, LLC

- Founded in 2005
- Staff
  - Four full-time ecologists
  - One part-time paleoecologist
  - Three full-time field technicians
  - Five summer interns
- Services
  - Science and planning
- Philosophy
  - Promote realistic planning
  - Assist, not direct

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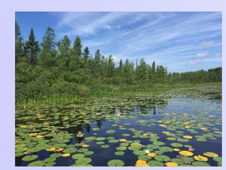
Why create a lake management plan?

- Preserve/restore ecological function to ensure cultural services
- To create a better understanding of lake's positive and negative attributes.
- To discover ways to minimize the negative attributes and maximize the positive attributes.
- Snapshot of lake's current status or health.
- Foster realistic expectations and dispel any misconceptions.



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Elements of an Effective Lake Management Planning Project

Data and Information Gathering

Environmental & Sociological

Planning Process

Brings it all together

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Data and Information Gathering

- **Study Components**
  - Water Quality Analysis
  - Paleocore Collection & Analysis
  - Watershed Assessment
  - Shoreland Assessment
  - Aquatic Plant Surveys
  - Fisheries data integration
  - Stakeholder Survey

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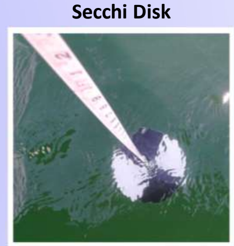
Water Quality Analysis

Phosphorus
Naturally occurring & essential for all life
Regulates phytoplankton biomass in most WI lakes
Most often 'limiting plant nutrient' (shortest supply)
Human development often increases P delivery to lakes

Chlorophyll-*a*
Pigment used in photosynthesis
Used as surrogate for phytoplankton biomass

Secchi Disk Transparency
Measure of water clarity
Measured using a Secchi disk

Secchi Disk



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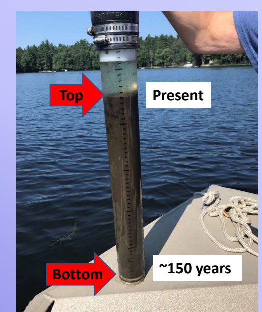
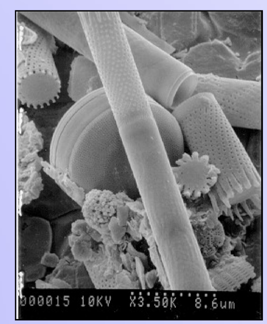
Paleocore Collection & Analysis



Sediment core



Diatoms

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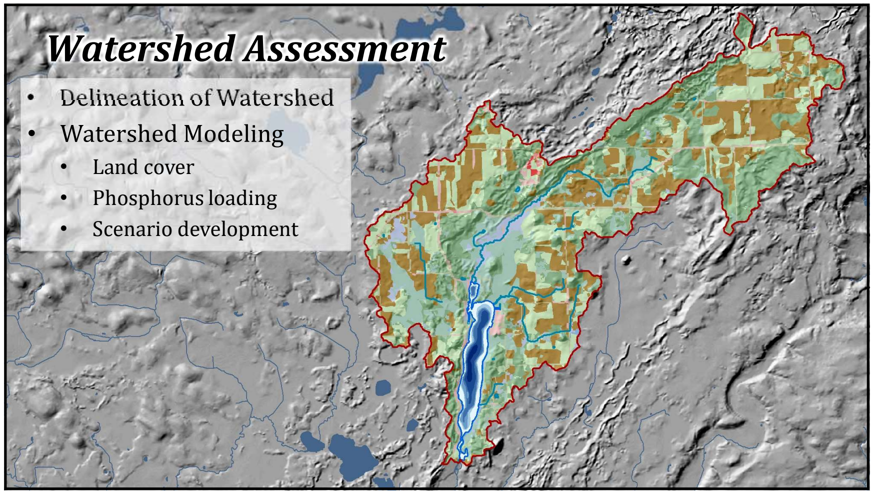
Watershed Assessment

- Delineation of Watershed
- Watershed Modeling
  - Land cover
  - Phosphorus loading
  - Scenario development



Shoreland Assessment

- Transition zone between land and water
- Important to maintain as much natural shoreline zone as possible

Completely Developed

Range

Completely Natural

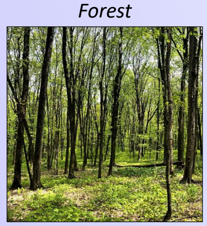
Native Aquatic Plants

- Foundation of the lake ecosystem
- Provide oxygen, food, and shelter
- Improve water quality
- Stabilize bottom and shoreline sediments

Lake

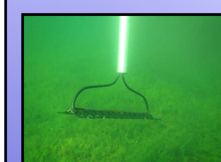
Grasslands

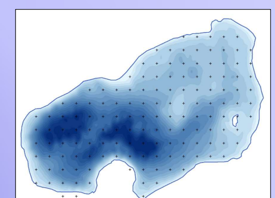
Forest



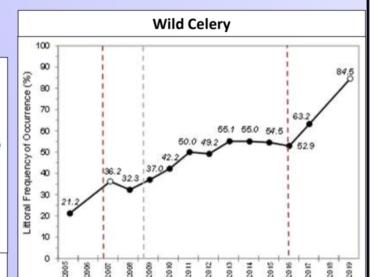
Point-Intercept Survey

- Grid-based survey (completed by WDNR)
- Determine abundance of each species
- Compare to other lakes
- Compare the same lake over time



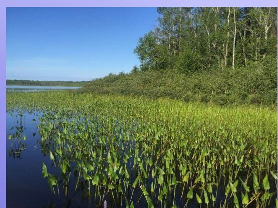


144 Sampling Locations
40-meter Resolution



Emergent & Floating-leaf Plant Communities

- Important communities for habitat, water quality, and shoreland stabilization
- Often negatively impacted by shoreland development



2010 Community Mapping Survey

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Non-Native Aquatic Plants

Eurasian Watermilfoil

Curly-leaf Pondweed
None Located

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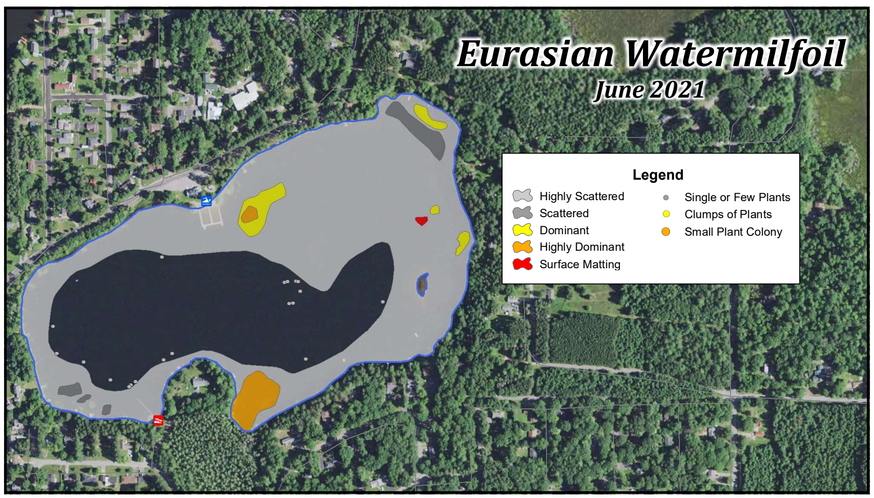

Non-Native Aquatic Plants

Pale-yellow Iris
None Located

Purple Loosestrife

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Fisheries Data Integration

- No fish sampling completed
- Assemble data from WDNR, USGS, & USFWS
- Fish survey results summaries (if available)
- Use information in planning as applicable

Stakeholder Survey

- Survey includes ERSLA members & riparian property owners
- Standard survey used as base
  - Planning committee potentially develops additional questions and options
  - Must not lead respondent to specific answer through a "loaded" question
- Survey must be approved by WDNR

Planning Process

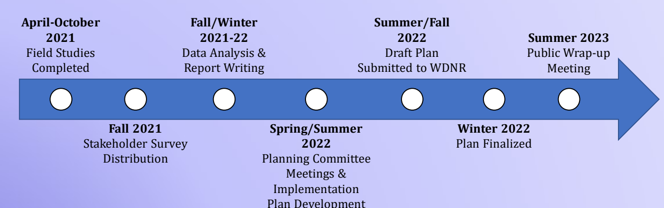
Planning Committee Meetings

Study Results (including a stakeholder survey)
Conclusions & Initial Recommendations
Management Goals
Management Actions
Timeframe
Facilitator(s)

↓

Implementation Plan

Project Timeline




Thank You


The Planning Process
…it's not as easy as you may think.
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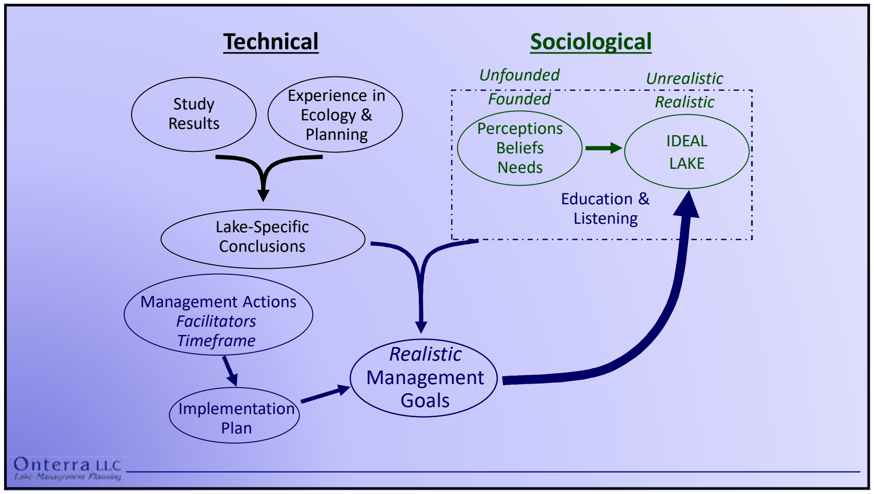
Technical
Sociological
Study Results
Experience in Ecology & Planning
Unfounded
Founded
Unrealistic
Realistic
Perceptions Beliefs Needs
IDEAL LAKE
Education & Listening
Lake-Specific Conclusions
Management Actions
Facilitators
Timeframe
Implementation Plan
Realistic Management Goals
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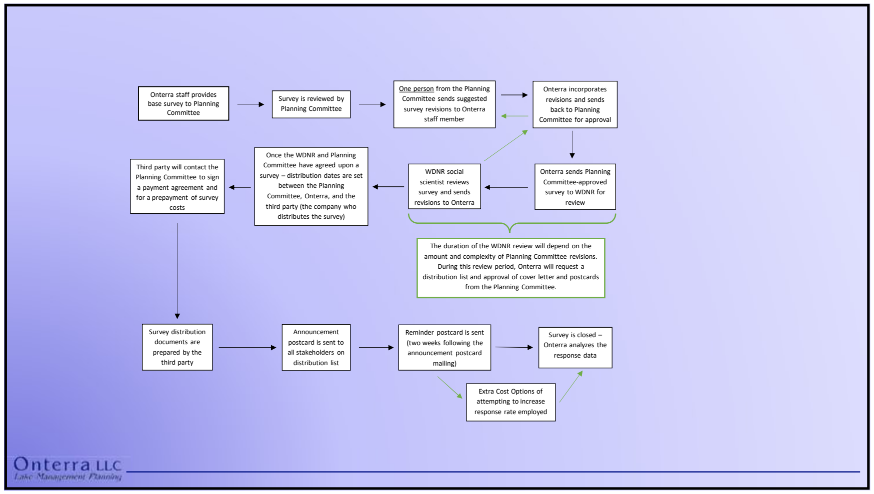
Onterra staff provides base survey to Planning Committee
Survey is reviewed by Planning Committee
One person from the Planning Committee sends suggested survey revisions to Onterra staff member
Onterra incorporates revisions and sends back to Planning Committee for approval
Onterra sends Planning Committee-approved survey to WDNR for review
WDNR social scientist reviews survey and sends revisions to Onterra
Once the WDNR and Planning Committee have agreed upon a survey – distribution dates are set between the Planning Committee, Onterra, and the third party (the company who distributes the survey)
Third party will contact the Planning Committee to sign a payment agreement and for a prepayment of survey costs
The duration of the WDNR review will depend on the amount and complexity of Planning Committee revisions. During this review period, Onterra will request a distribution list and approval of cover letter and postcards from the Planning Committee.
Survey distribution documents are prepared by the third party
Announcement postcard is sent to all stakeholders on distribution list
Reminder postcard is sent (two weeks following the announcement postcard mailing)
Survey is closed – Onterra analyzes the response data
Extra Cost Options of attempting to increase response rate employed
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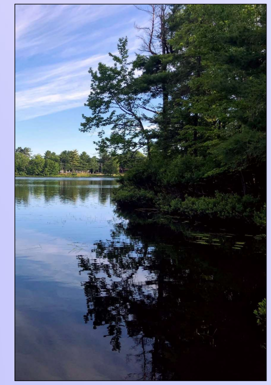
Management Planning Project Overview

- Current project designed to assess the overall status of the lake & update management plan
- Collect & analyze data – completed
  - Technical & sociological
- Construct long-term & useable plan



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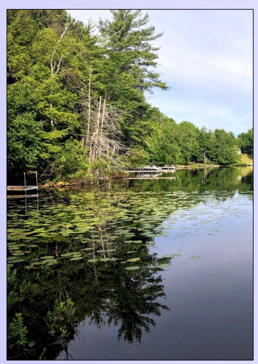
Planning I Meeting Agenda

- Management Planning Project Overview
- Study Results
  - Water Quality
  - Watershed
  - Shoreland Condition
  - Aquatic Plants
  - Fisheries Data Integration
  - Conservation Opportunity Areas
- "Big Picture" Conclusions
- Planning Meeting II: AIS Management & Goal Development



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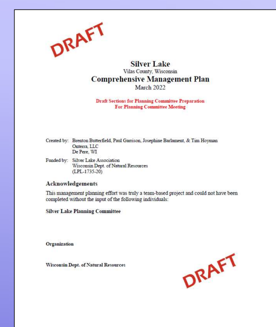
Management Plan Outline



Planning Meeting I

Planning Meeting II

- 1.0 Introduction
- 2.0 Stakeholder Participation
- 3.0 Study Results
  - 3.1 Water Quality
  - 3.2 Watershed Assessment
  - 3.3 Shoreland Condition
  - 3.4 Aquatic Plants
  - 3.5 Aquatic Invasive Species
  - 3.6 Fisheries Data Integration
  - 3.7 Areas of Special Conservation Interest
- 4.0 Summary & Conclusions
- 5.0 Implementation Plan
- 6.0 Methods
- 7.0 Literature Cited

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Summary of General Project Results

Water Quality

- Overall, water quality is *excellent* for a deep headwater drainage lake in Wisconsin
- Lake has low nutrient/algae concentrations and high water clarity

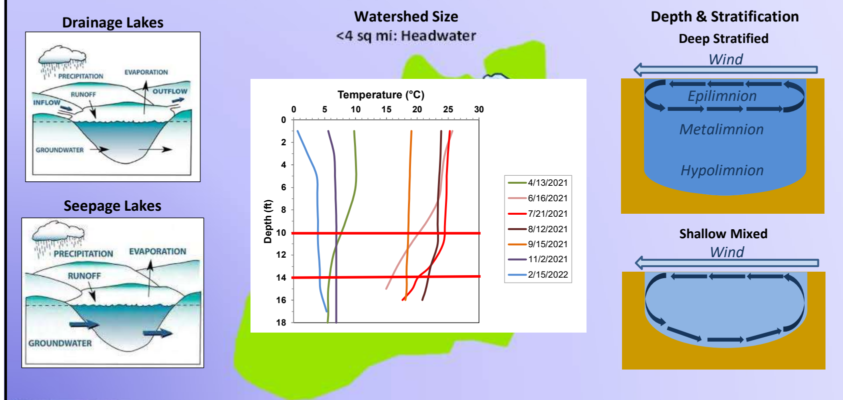
Watershed & Immediate Shoreline

- ~35% of watershed is developed (urban); however, wetlands likely buffering lake from impacts of urban runoff
- Majority of shoreland zone has minimal development & contains intact natural habitat

Aquatic Plant Community

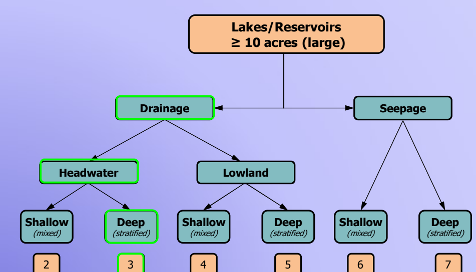
- Lake supports a high-quality native aquatic plant community – high number of native species and number of sensitive species
- Plant community has seen changes in abundance of certain species between 2005-2021
- Eurasian watermilfoil population at highest occurrence in 2021 since surveys began in 2005
- A few purple loosestrife occurrences were mapped along the northern shore in 2021

Wisconsin Lakes Natural Community Types



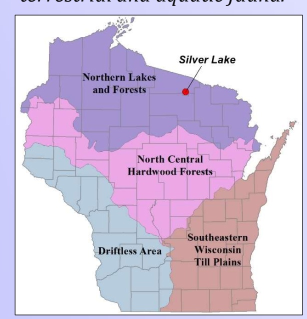
Natural Community Types

Categorization of lakes with similar features that influence water quality



Ecoregions

An area containing similar geology, physiography, hydrology, climate, and soils. As well as common terrestrial and aquatic fauna.



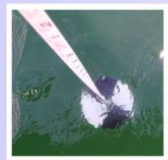
Introduction to Lake Water Quality

Phosphorus
Naturally occurring & essential for all life
Regulates phytoplankton biomass in most WI lakes
Most often 'limiting plant nutrient' (shortest supply)
Human development often increases P delivery to lakes

N:P Ratio: 30:1 – Phosphorus

Chlorophyll-*a*
Pigment used in photosynthesis
Used as surrogate for phytoplankton biomass

Secchi Disk Transparency
Measure of water clarity
Measured using a Secchi disk



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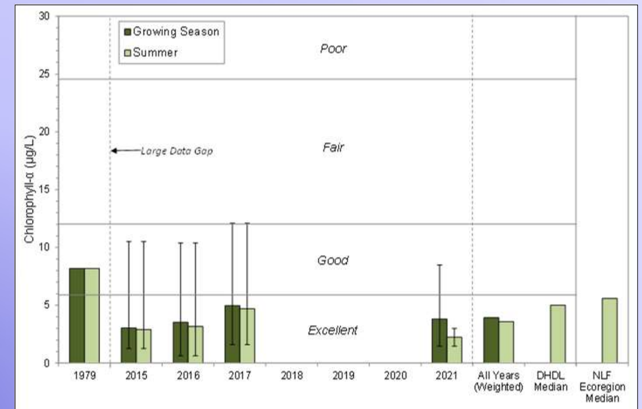
Chlorophyll-α



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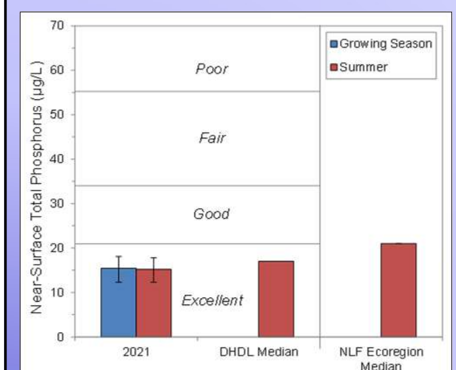
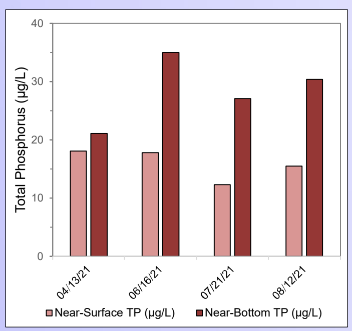
Total Phosphorus





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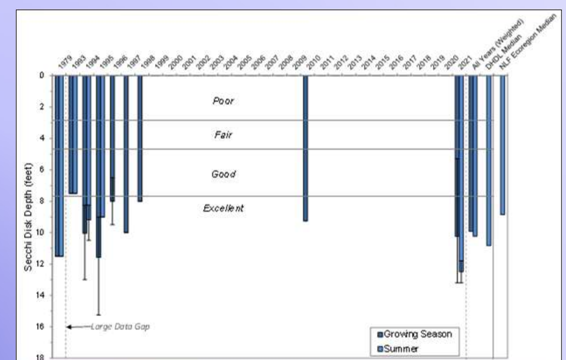
Secchi Disk Depth



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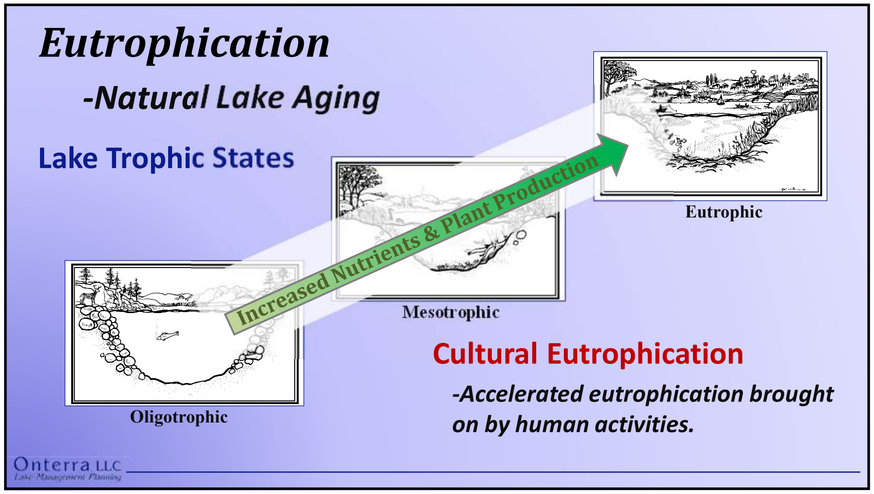
Eutrophication
-Natural Lake Aging
Lake Trophic States
Increased Nutrients & Plant Production
Eutrophic
Mesotrophic
Oligotrophic
Cultural Eutrophication
-Accelerated eutrophication brought on by human activities.
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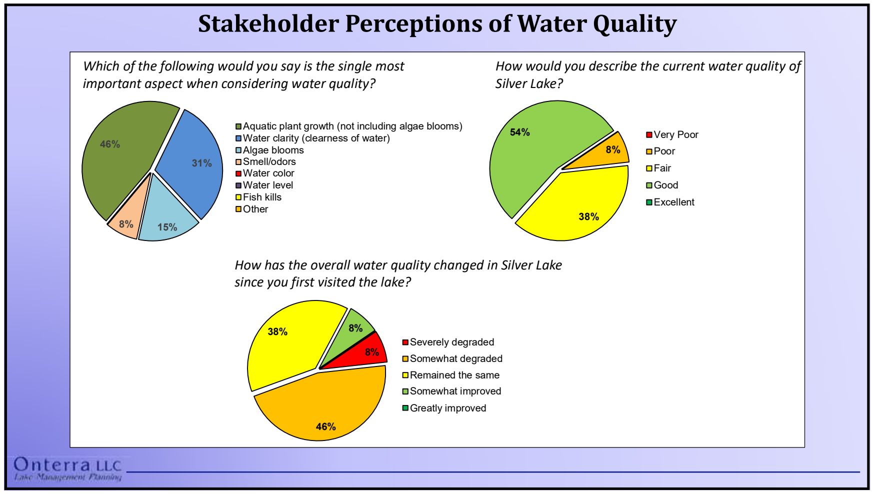
Stakeholder Perceptions of Water Quality
Which of the following would you say is the single most important aspect when considering water quality?
46%
31%
15%
8%
Aquatic plant growth (not including algae blooms)
Water clarity (clearness of water)
Algae blooms
Smell/odors
Water color
Water level
Fish kills
Other
How would you describe the current water quality of Silver Lake?
54%
8%
38%
Very Poor
Poor
Fair
Good
Excellent
How has the overall water quality changed in Silver Lake since you first visited the lake?
38%
8%
8%
46%
Severely degraded
Somewhat degraded
Remained the same
Somewhat improved
Greatly improved
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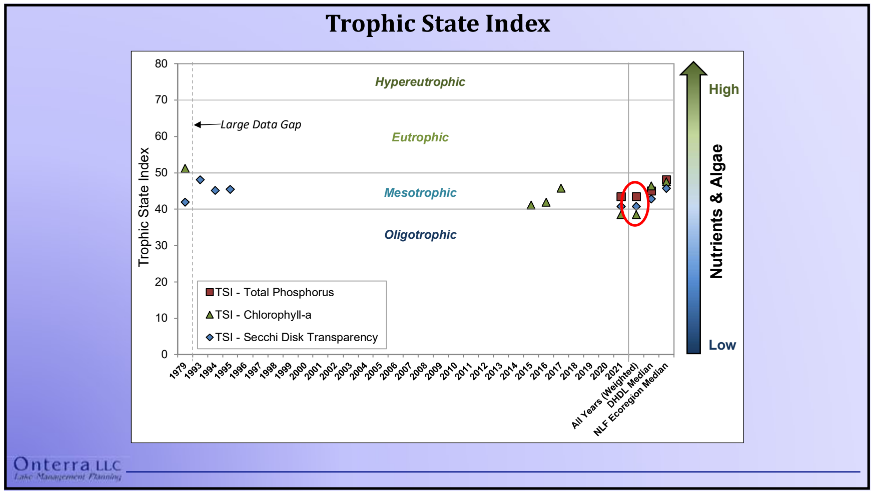
Trophic State Index
Hypereutrophic
Eutrophic
Mesotrophic
Oligotrophic
Large Data Gap
Trophic State Index
TSI - Total Phosphorus
TSI - Chlorophyll-a
TSI - Secchi Disk Transparency
All Years (Weighted)
DHDL Median
NLF Ecoregion Median
High
Nutrients & Algae
Low
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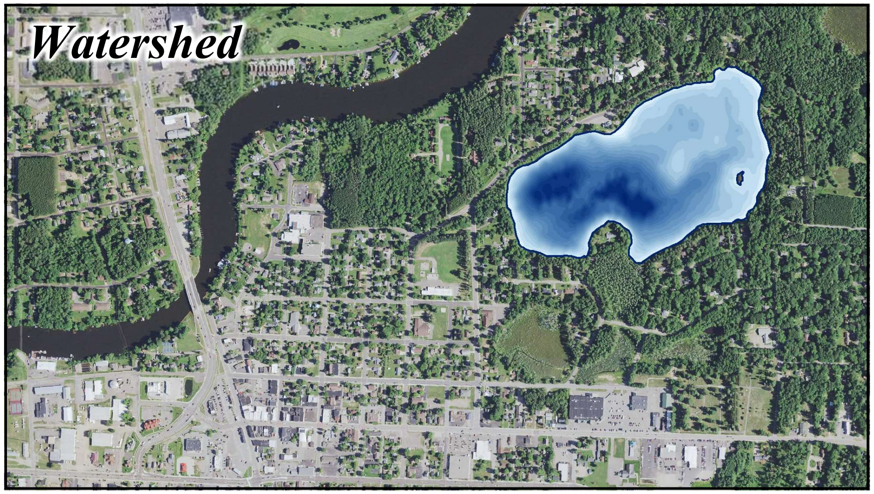
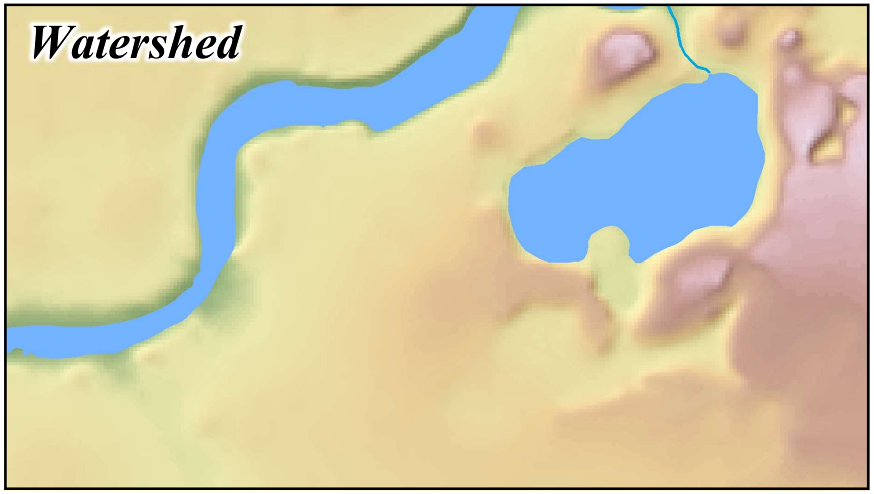
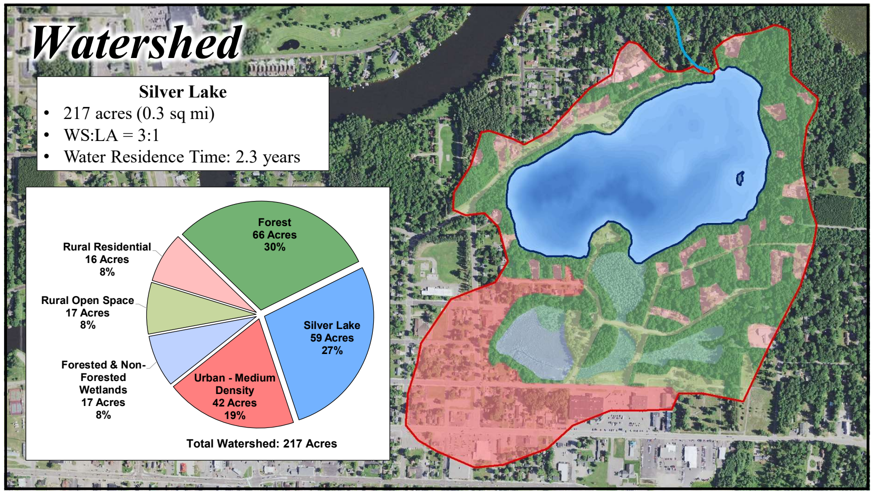
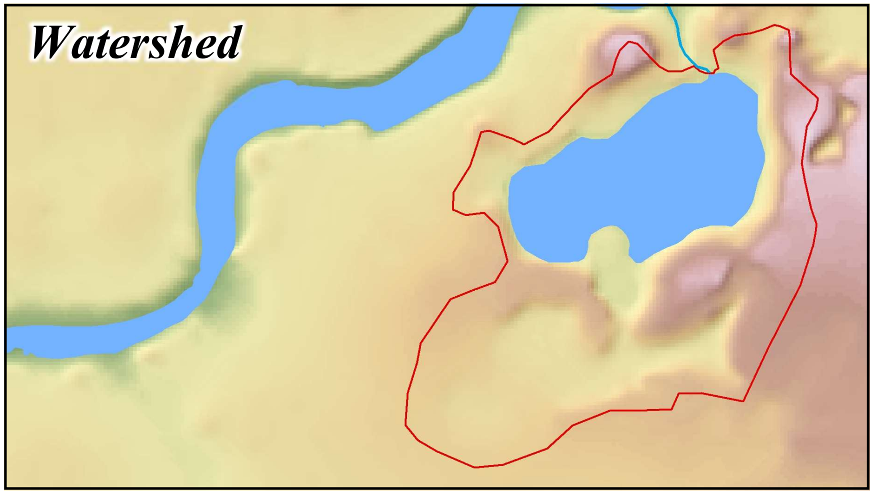

Watershed







Phosphorus Loading

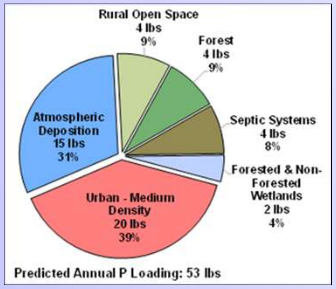
WiLMS Predicted Annual Phosphorus Load from Watershed: **53 pounds**

Predicted In-Lake Growing Season TP Concentration: **30 μg/L**

Measured In-Lake Growing Season TP Concentration: **16 μg/L**

Phosphorus loading is **overestimated** by model

Actual annual load likely closer to **<20 pounds (>60% lower than predicted)**



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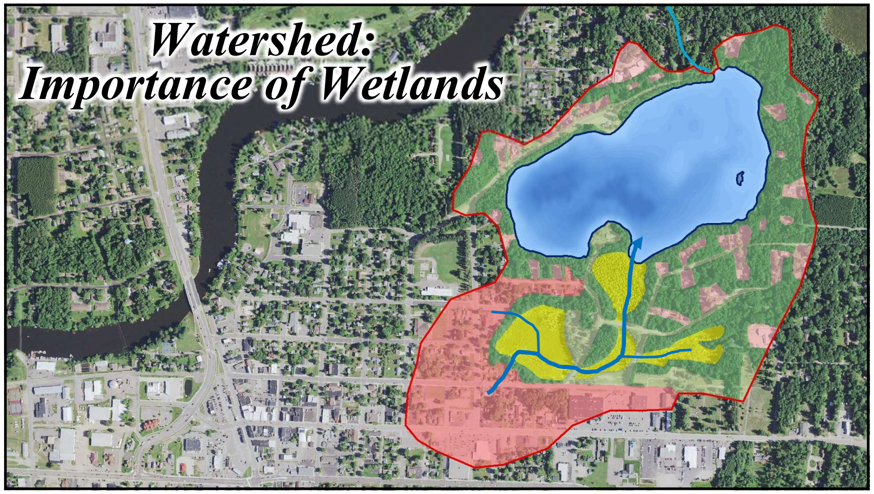
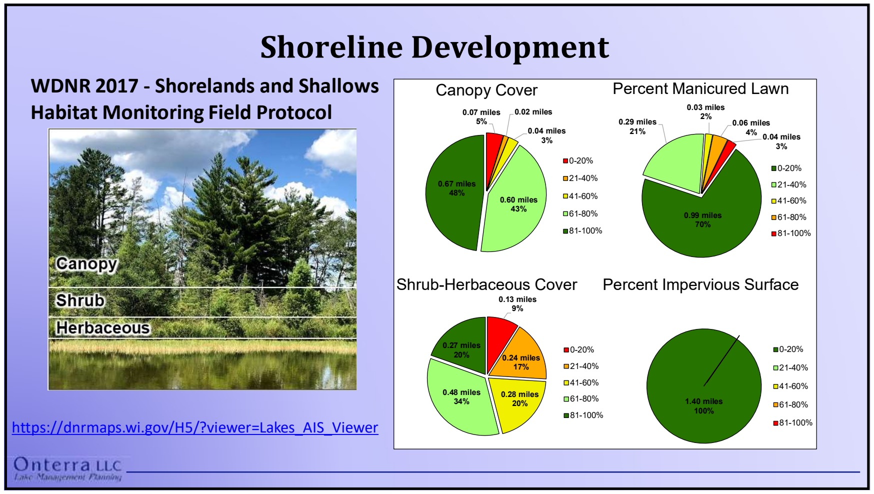

Watershed:
Importance of Wetlands


Shoreline Development
WDNR 2017 - Shorelands and Shallows Habitat Monitoring Field Protocol
Canopy
Shrub
Herbaceous
https://dnrmaps.wi.gov/H5/?viewer=Lakes_AIS_Viewer
Canopy Cover
0.07 miles 5%
0.02 miles
0.04 miles 3%
0.67 miles 48%
0.60 miles 43%
0-20%
21-40%
41-60%
61-80%
81-100%
Percent Manicured Lawn
0.03 miles 2%
0.29 miles 21%
0.06 miles 4%
0.04 miles 3%
0.99 miles 70%
0-20%
21-40%
41-60%
61-80%
81-100%
Shrub-Herbaceous Cover
0.13 miles 9%
0.27 miles 20%
0.24 miles 17%
0.48 miles 34%
0.28 miles 20%
0-20%
21-40%
41-60%
61-80%
81-100%
Percent Impervious Surface
1.40 miles 100%
0-20%
21-40%
41-60%
61-80%
81-100%
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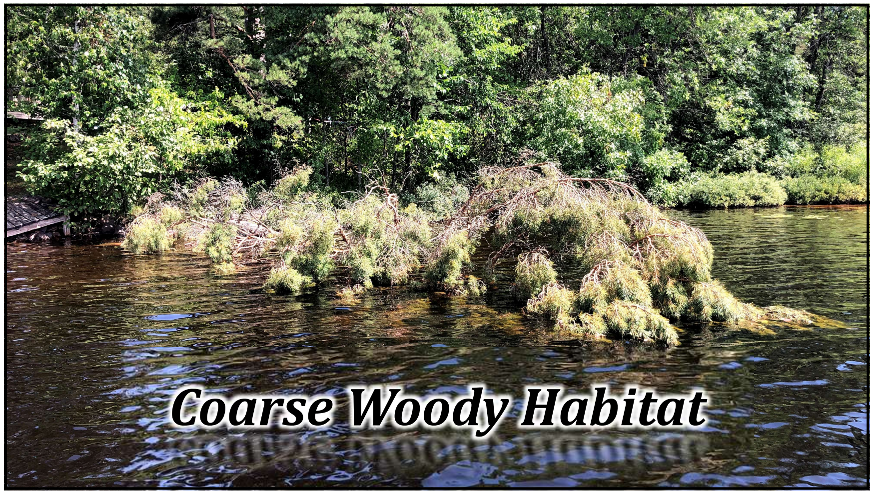
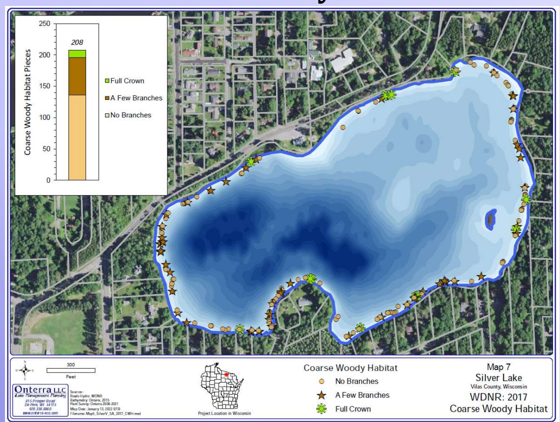
Shoreland Condition


Coarse Woody Habitat

Coarse Woody Habitat



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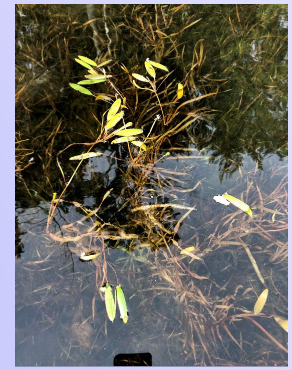
Aquatic Plant Surveys

- Assess both non-native & native species
- Four surveys completed in 2021
  - Early-Season AIS Survey
  - Whole-lake Point-Intercept Survey
  - Emergent/Floating-leaf Community Mapping Survey
  - Late-Season AIS Survey



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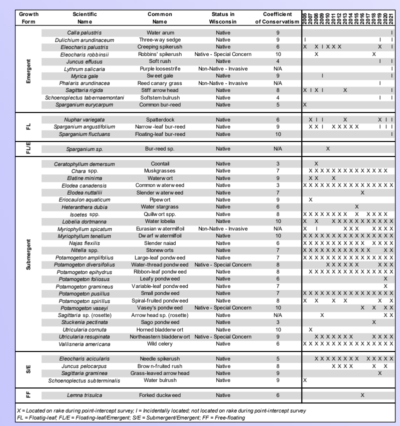
Plant Data Overview

- 44 native plant species recorded from 2005-2021 surveys
- 4 listed as 'special concern'
- 4 non-native species
  - Eurasian watermilfoil
  - Purple loosestrife
  - Narrow-leaved Cattail
  - Reed canary grass
- Max Rooting Depth in 2021: 16'



Growth Form	Scientific Name	Common Name	Status in Wisconsin	Coefficient of Conservatism	2005	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Emergent	*Calla palustris*	Water arum	Native	9																I
	Dulichium arundinaceum	Three-way sedge	Native	9	I													I		I
	Eleocharis palustris	Creeping spikerush	Native	6	X		X	I	X	X	X							X		I
	Eleocharis robbinsii	Robbins' spikerush	Native - Special Concern	10			X										X			
	Juncus effusus	Soft rush	Native	4														I		I
	Lythrum salicaria	Purple loosestrife	Non-Native - Invasive	N/A																I
	Myrica gale	Sweet gale	Native	9				I												I
	Phalaris arundinacea	Reed canary grass	Non-Native - Invasive	N/A														I		
	Sagittaria rigida	Stiff arrowhead	Native	8	X	I	X	I				X								I
	Schoenoplectus tabernaemontani	Softstem bulrush	Native	4														I		I
	Sparganium eurycarpum	Common bur-reed	Native	5	X															
FL	*Nuphar variegata*	Spatterdock	Native	6		X	I	I				X						X	I	I
	Sparganium angustifolium	Narrow-leaf bur-reed	Native	9		X	X	I		X	X	X	X	X				I	I	I
	Sparganium fluctuans	Floating-leaf bur-reed	Native	10																I
FL/E	*Sparganium sp.*	Bur-reed sp.	Native	N/A					X											
Submergent	*Ceratophyllum demersum*	Coontail	Native	3			X													
	Chara spp.	Muskgrasses	Native	7		X	X	X	X	X	X	X	X	X	X	X	X	X	X	
	Elatine minima	Waterwort	Native	9		X	X			X										
	Elodea canadensis	Common waterweed	Native	3	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Elodea nuttallii	Slender waterweed	Native	7											X					
	Eriocaulon aquaticum	Pipewort	Native	9		X														
	Heteranthera dubia	Water stargrass	Native	6										X						
	Isoetes spp.	Quillwort spp.	Native	8	X	X	X	X	X	X	X	X		X		X	X	X	X	
	Lobelia dortmanna	Water lobelia	Native	10	X		X			X	X	X	X	X	X	X	X	X	X	X
	Myriophyllum spicatum	Eurasian watermilfoil	Non-Native - Invasive	N/A	X		I					X	X	X			X	X	X	X
	Myriophyllum tenellum	Dwarf watermilfoil	Native	10	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Najas flexilis	Slender naiad	Native	6	X	X	X	X	X	X	X	X	X	X	X		X	X	X	X
	Nitella spp.	Stoneworts	Native	7	X	X	X	X	X	X	X	X	X	X	X	X			X	X
	Potamogeton amplifolius	Large-leaf pondweed	Native	7	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Potamogeton diversifolius	Water-thread pondweed	Native - Special Concern	8					X	X	X	X	X	X				X	X	X
	Potamogeton epihydrus	Ribbon-leaf pondweed	Native	8		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Potamogeton foliosus	Leafy pondweed	Native	6															X	
	Potamogeton gramineus	Variable-leaf pondweed	Native	7															X	
	Potamogeton pusillus	Small pondweed	Native	7	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Potamogeton spirillus	Spiral-fruited pondweed	Native	8	X		X			X		X						X		X
	Potamogeton vaseyi	Vasey's pondweed	Native - Special Concern	10											X		X		X	X
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	Native	N/A				X											X	X
	Stuckenia pectinata	Sago pondweed	Native	3													X			
	Utricularia cornuta	Horned bladderwort	Native	10		X														
	Utricularia resupinata	Northeastern bladderwort	Native - Special Concern	9			X	X	X	X	X	X	X				X	X	X	X
	Vallisneria americana	Wild celery	Native	6	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
S/E	*Eleocharis acicularis*	Needle spikerush	Native	5			X	X	X	X	X	X	X	X		X	X	X	X	X
	Juncus pelocarpus	Brown-fruited rush	Native	8						X	X	X	X				X		X	
	Sagittaria graminea	Grass-leaved arrowhead	Native	9														X		
	Schoenoplectus subterminalis	Water bulrush	Native	9	X															
FF	*Lemna trisulca*	Forked duckweed	Native	6											X					

X = Located on rake during point-intercept survey; I = Incidentally located; not located on rake during point-intercept survey
FL = Floatig-leaf; FL/E = Floating-leaf/Emergent; S/E = Submergent/Emergent; FF = Free-floating

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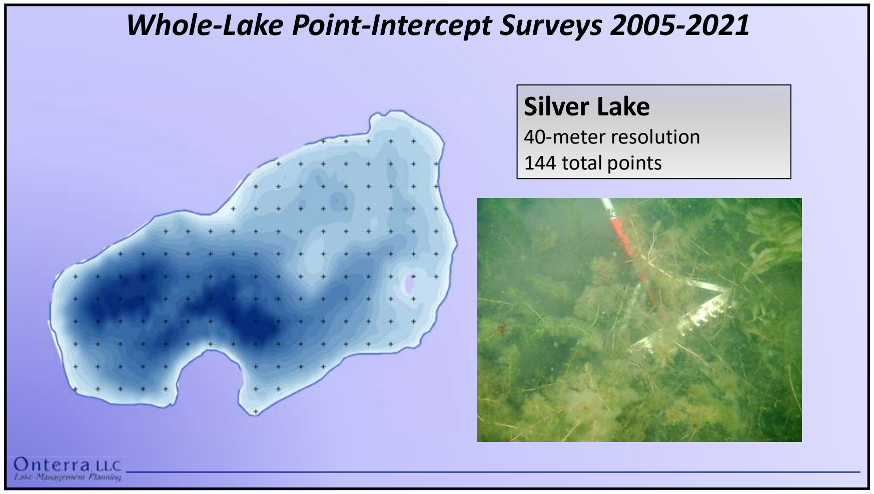
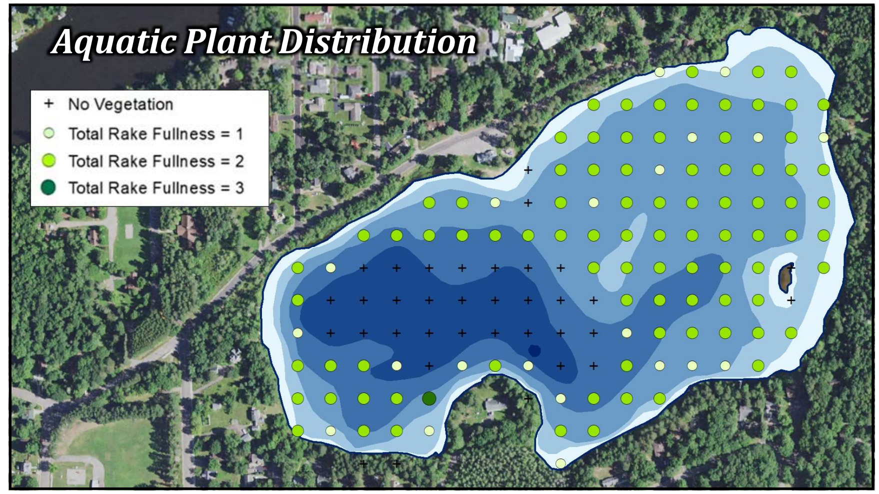
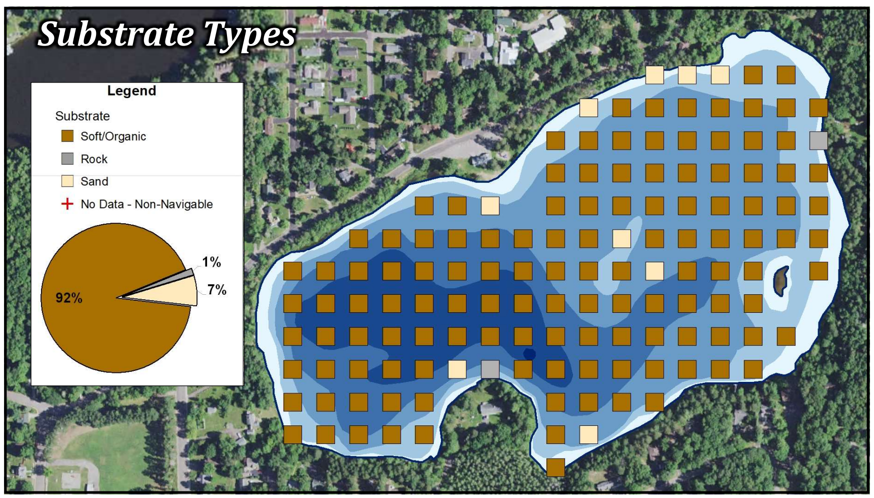
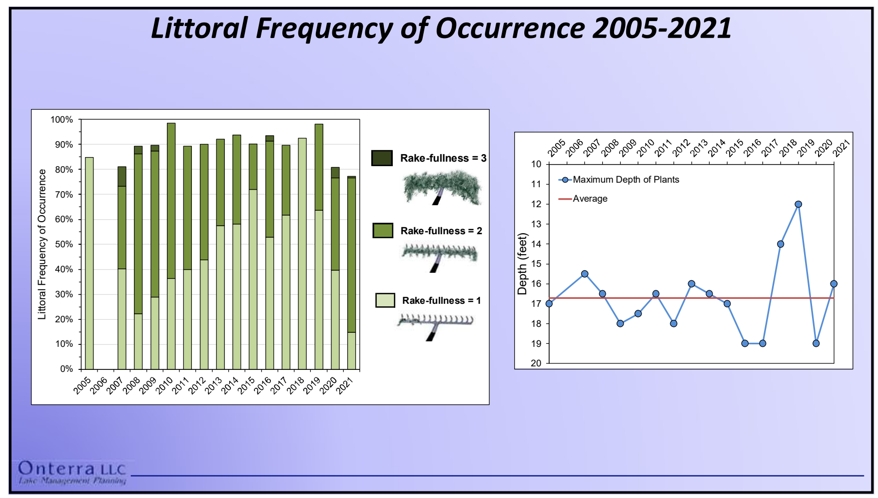

Whole-Lake Point-Intercept Surveys 2005-2021
Silver Lake
40-meter resolution
144 total points
Onterra LLC
Lake Management Planning


Aquatic Plant Distribution
No Vegetation
Total Rake Fullness = 1
Total Rake Fullness = 2
Total Rake Fullness = 3


Substrate Types
Legend
Substrate
Soft/Organic
Rock
Sand
No Data - Non-Navigable
92%
1%
7%


Littoral Frequency of Occurrence 2005-2021
Littoral Frequency of Occurrence
Rake-fullness = 3
Rake-fullness = 2
Rake-fullness = 1
Maximum Depth of Plants
Average
Depth (feet)
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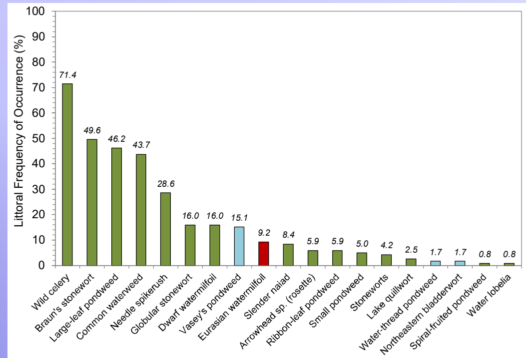
2021 Littoral Frequency of Occurrence



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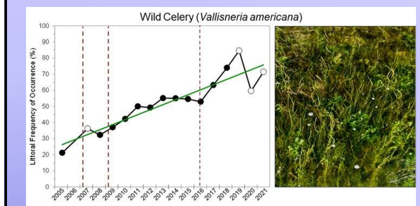
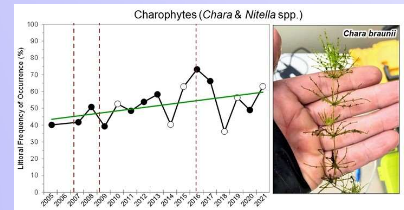
Species' Changes in Occurrence





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Rare Species

Water-thread Pondweed
(*Potamogeton diversifolius*)

Vasey's Pondweed
(*Potamogeton vaseyi*)

Robbins' Spikerush
(*Eleocharis robbinsii*)

Northeastern Bladderwort
(*Utricularia resupinata*)

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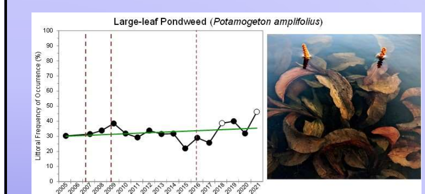
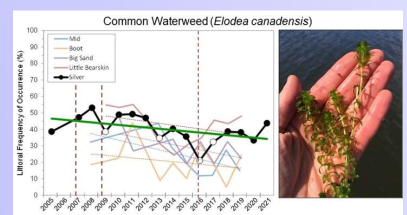
Species' Changes in Occurrence





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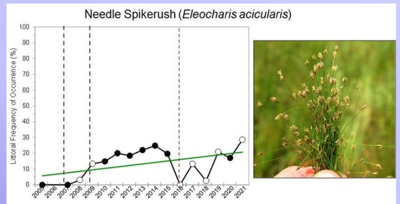
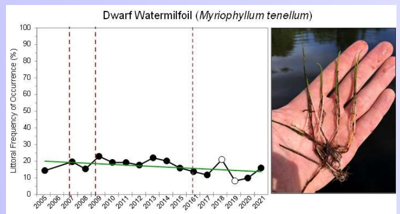
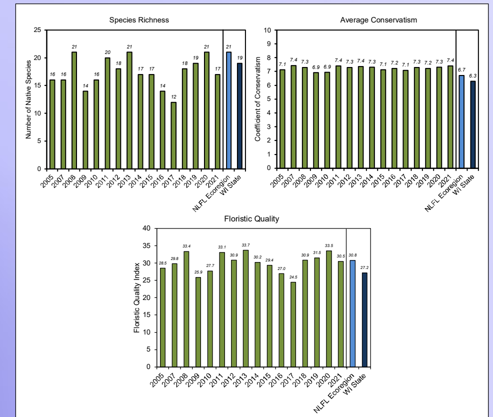
Species' Changes in Occurrence

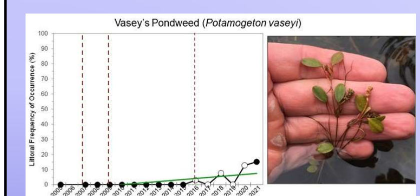


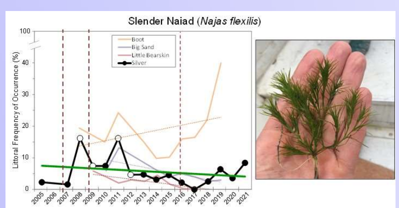


Floristic Quality Analysis

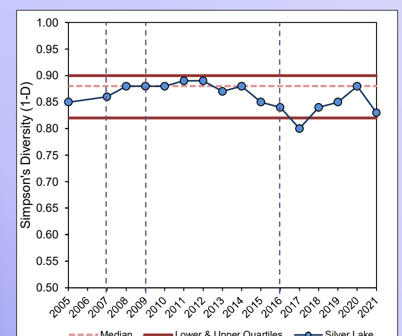


Species' Changes in Occurrence

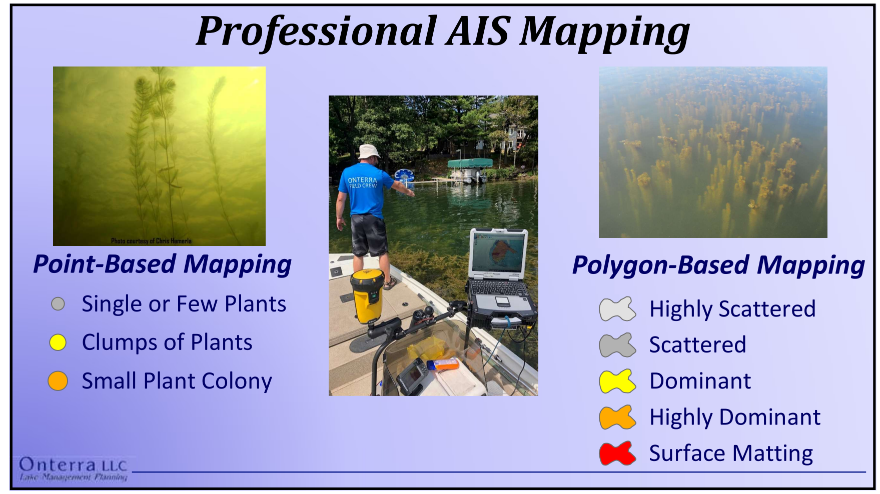




Simpson's Diversity Index




Emergent & Floating-leaf Aquatic Plants


Professional AIS Mapping
Point-Based Mapping
Single or Few Plants
Clumps of Plants
Small Plant Colony
Polygon-Based Mapping
Highly Scattered
Scattered
Dominant
Highly Dominant
Surface Matting
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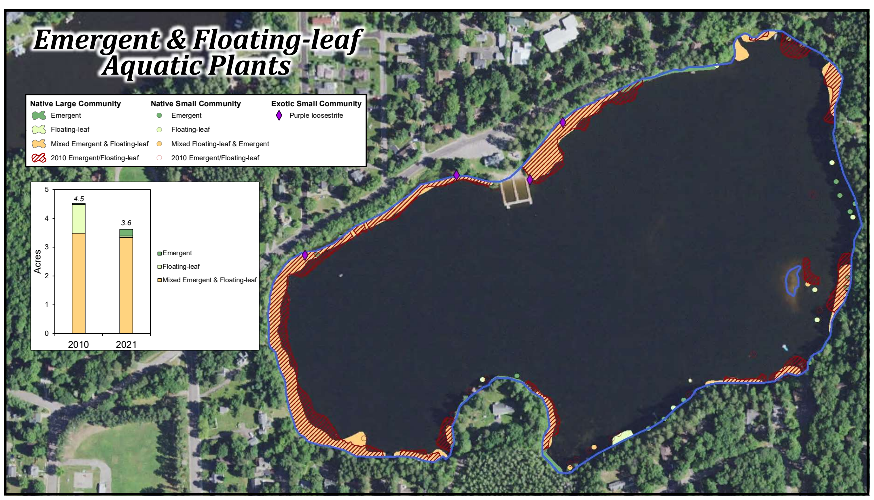
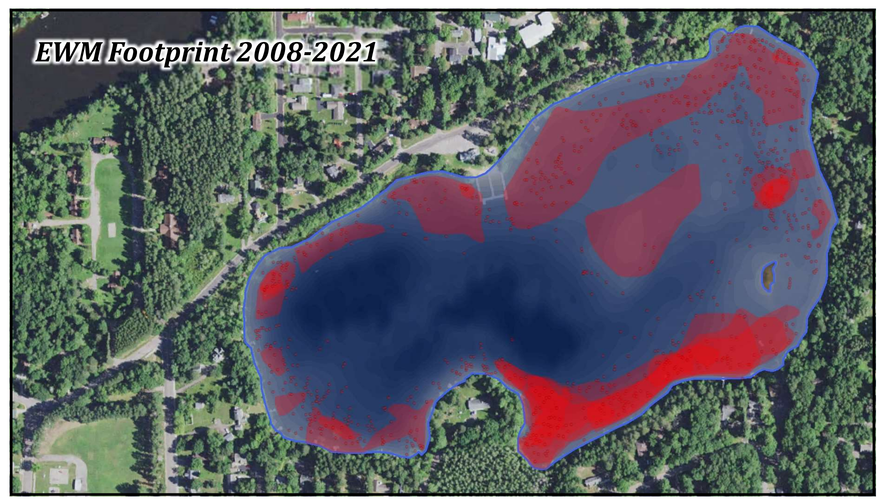
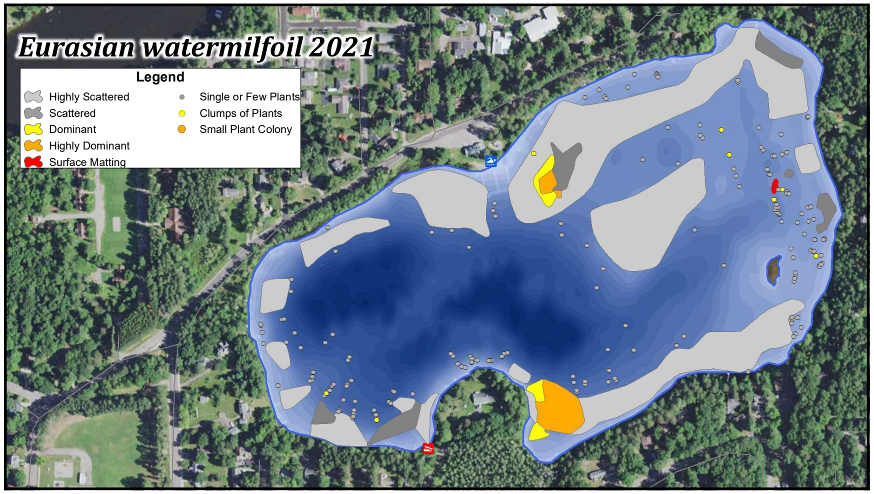
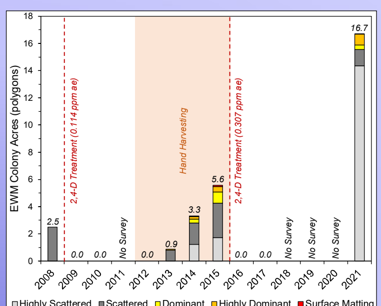
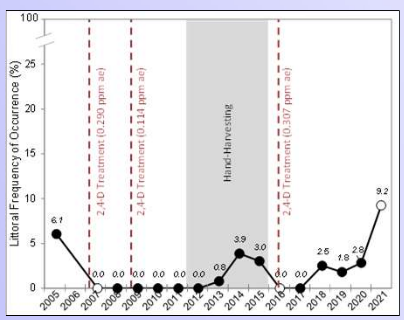

Emergent & Floating-leaf Aquatic Plants
Native Large Community
Emergent
Floating-leaf
Mixed Emergent & Floating-leaf
2010 Emergent/Floating-leaf
Native Small Community
Emergent
Floating-leaf
Mixed Floating-leaf & Emergent
2010 Emergent/Floating-leaf
Exotic Small Community
Purple loosestrife
Acres
0
1
2
3
4
5
4.5
3.6
2010
2021
Emergent
Floating-leaf
Mixed Emergent & Floating-leaf


EWM Footprint 2008-2021



EWM 2008-2021





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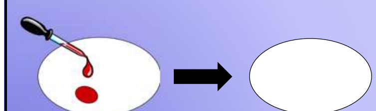
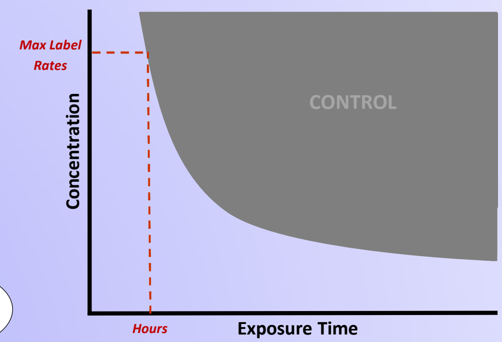
Ecological Definitions of Herbicide Treatment

Spot Treatment:

Herbicide applied at a scale where dissipation will not result in significant lake wide concentrations; impacts are anticipated to be localized to in/around application area.





High Concentration ► Short Exposure Time

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Horizontal Herbicide Mixing (Dissipation)

- ~25 acres of 305 acre lake (8%)
- Tracer Dye (Rhodamine WT) Survey

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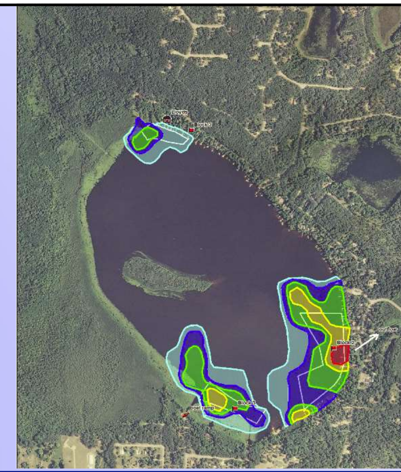
2.5 HAT

12/24 hours for mortality { 75-100% 50-75%
25-50%
10-25%
5-10%



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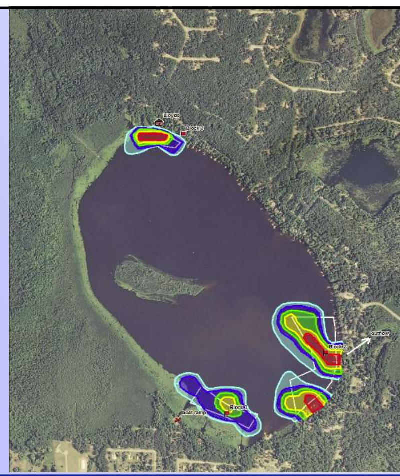
1 HAT

12/24 hours for mortality { 75-100% 50-75%
25-50%
10-25%
5-10%



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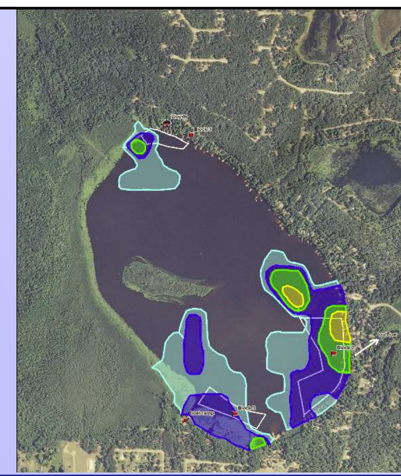
Lake Management Planning

4 HAT

12/24 hours for mortality { 75-100% 50-75%
25-50%
10-25%
5-10%



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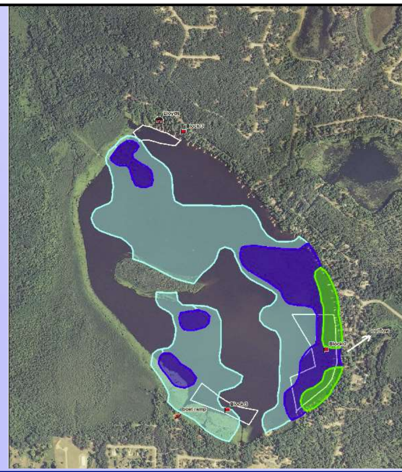
Lake Management Planning

6 HAT

12/24 hours for mortality { 75-100% 50-75%
25-50%
10-25%
5-10%

2,4-D CET needed for EWM control based upon published studies:
- **sustained 4.0 ppm for 12 hours**
- **sustained 2.0 ppm for 24 hours**
- **0.1-0.3 ppm for 6 weeks**



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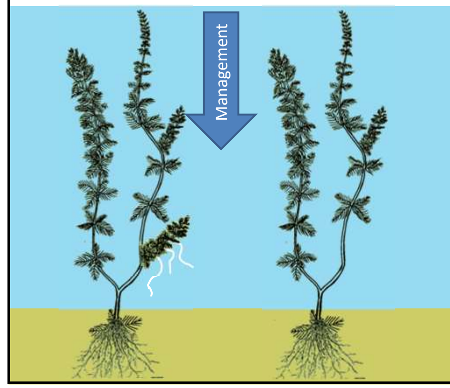
EWM Life-Cycle & Control Strategy Philosophy

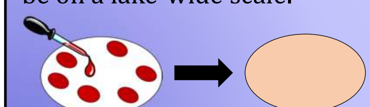


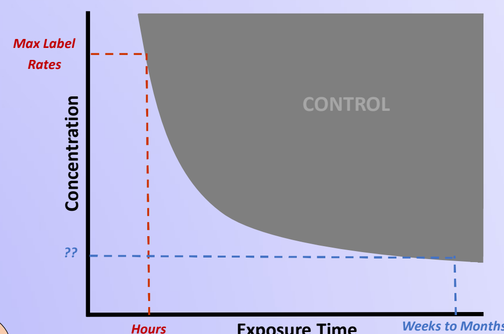
- Herbicide needs to translocate to root crown (*hard to kill with herbicides*)
- Hand-harvesting that extracts roots is effective (*extremely time intensive*)
- Mechanical harvesting can minimize nuisance conditions (*spread to new areas not a concern for established populations)*
- Sometimes EWM does not cause nuisance conditions or ecological changes

Ecological Definitions of Herbicide Treatment

Whole-Lake/Basin Treatment:

Herbicide applied at a scale where dissipation will result in significant lake wide concentrations; impacts are anticipated to be on a lake-wide scale.





Low Concentration ▶ Long Exposure Time

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Spot Treatment Guidance

- Actual CET in the field is more difficult to predict and maintain in spot treatments due to **<u>dissipation</u>**
- Rapid dissipation of herbicide occurs in 1-6 HAT in many (most?) spot-treatments
- Size (large vs small), shape (broad vs thin/linear), and location (protected vs exposed) matters
- Achieving EWM population suppression for at least 2 summers is definition of success

Onterra LLC Lake Management Planning

AIS Management Perspectives

1. **No Coordinated Active Management (Let Nature Take its Course)**
   - Lake group does not lead efforts
   - Encourage nuisance abatement through manual removal by property owners
2. **Minimize navigation and recreation impediment (Nuisance Mgmt)**
   - May be accomplished through mechanical harvesting or herbicide treatment
   - Prioritize areas based on human use & HWM density
3. **Reduce AIS Population on a lake-wide level (Population Management)**
   - Most applicable for new discoveries, whole-lake herbicide, water level drawdown
   - Not possible on some systems with current management "toolbox"
   - Will not eradicate AIS
   - Set triggers (thresholds) of implementation and tolerance

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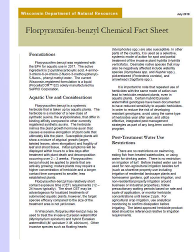
Florpyrauxifen-benzyl (ProcellaCOR™)

- New class of synthetic auxin hormone mimics
  - Much different binding affinity than other auxins
  - Use at PPB rate vs PPM
- Short contact exposure time (CET) requirement
- Short environmental fate
  - Half life 1-6 days (photolysis, higher rates in clear water)
  - High Koc (soil/organic binding affinity)
- Currently formulated for spot treatments, but manufacturer working towards whole-lake use patterns
- Detailed information on field applications is limited


Florpyrauxifen-benzyl Chemical Fact Sheet

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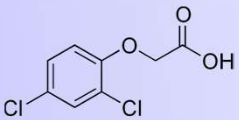
2.4-D Impacts on Early Fish Life Stages

- DeQuattro and Karasov 2016 demonstrated statistically valid reduction in fathead minnow larval survivability when 2,4-D is exposed to embryo (eggs) and larval (hatched). Also demonstrated sub-lethal endocrine disruption impacts (tubercles).
- Dehnert et. al 2018 indicates the first 14 days post hatch (dph) is the most critical period for fathead minnow.
- Dehnert et. al 2021 investigated multiple gamefish species, exposing to 30 dph to conform with EPA's definition of "chronic"



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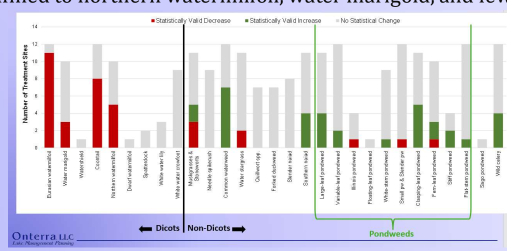
ProcellaCOR™ 2019-2021 Field Trials

- Onterra has monitored dozens of treatments
- Nearly all show high level of initial control with little to no EWM/HWM extending through year-after-treatment
- Dissipation & mixing resulting in off-target impacts on many projects
- Slightly reduced efficacy and dissipation in high pH lakes, SE WI lakes
- Native plant impacts largely confined to northern watermilfoil, water marigold, and few other select dicots



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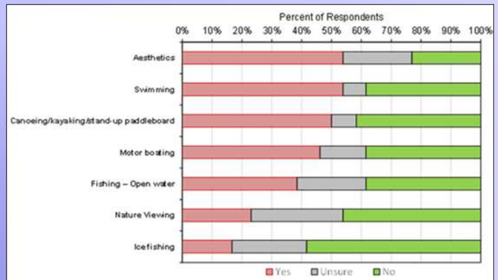
Stakeholder Survey – Invasive Plant Management

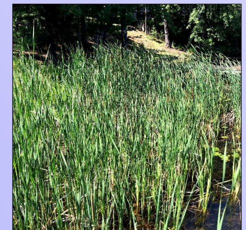
Has the EWM population ever had a negative impact on your enjoyment of Silver Lake?



Shoreland Invasive Plants

Purple Loosestrife

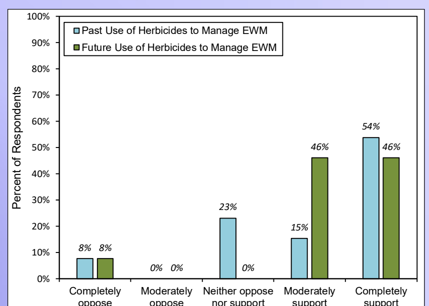
Narrow-leaved Cattail



Reed Canary Grass

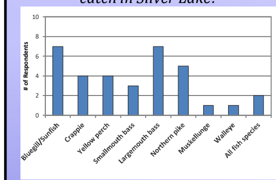
Stakeholder Survey – Invasive Plant Management

What is your level of support or opposition for the past and future use of aquatic herbicides to target EWM in Silver Lake?



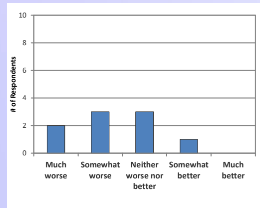
Fisheries – Stakeholder Survey

What species of fish do you like to catch in Silver Lake?



How would you describe the current quality of fishing on Silver Lake?

How has the quality of fishing changed on Silver Lake since you have started fishing the lake?



Fisheries – Population Trends

Muskellunge
- Not captured in 2015 survey but are present in low abundance
- Population is self-sustaining

Largemouth Bass
- Most common gamefish in Silver Lake

Northern Pike
- Present, but 2015 survey methods did not allow for population estimate

Walleye
- Previously stocked, but stocking has not occurred since 1940
- Four walleyes captured during 2015 survey – like illegally transferred to Silver Lake

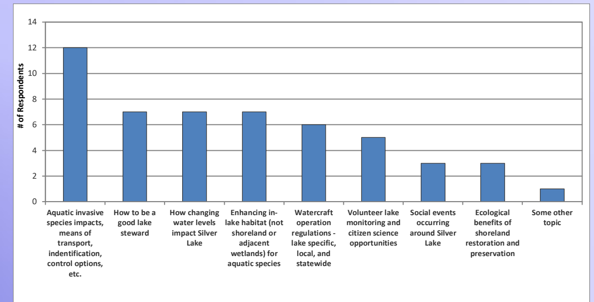
Panfish
- Bluegill most encountered followed by pumpkinseed
- Yellow perch also present

Silver Lake Areas of Special Conservation Interest (ASCIs)

- ASCIs delineated in Silver Lake based on aquatic plant community & shoreland assessments
    - Intent is to encompass and highlight the full spectrum of native species and natural community diversity present in Silver Lake
    - Capture areas of highest plant species richness and diversity
    - Proximity to natural shorelines and coarse woody habitat abundance
    - Nearly encompasses entirety of Silver Lake's riparian zone

Stakeholder Survey Results

Which of these subjects would you like to learn more about?



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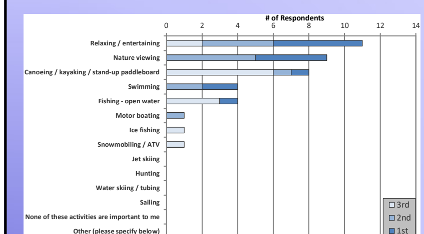
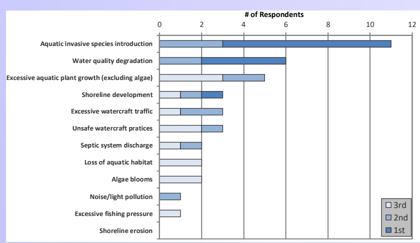
Stakeholder Survey Results

Rank up to three activities that are important reasons for owning property on or near Silver Lake.



Please rank your top three concerns regarding Silver Lake.



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Big Picture Conclusions

Water Quality

- Overall, water quality is *excellent* for a deep headwater drainage lake in Wisconsin
- Lake has low nutrient/algae concentrations and high water clarity

Watershed & Immediate Shoreline

- ~35% of watershed is developed (urban); however, wetlands likely buffering lake from impacts of urban runoff
- Majority of shoreland zone has minimal development & contains intact natural habitat

Aquatic Plant Community

- Lake supports a high-quality native aquatic plant community – high number of native species and number of sensitive species
- Plant community has seen changes in abundance of certain species between 2005-2021
- Eurasian watermilfoil population at highest occurrence in 2021 since surveys began in 2005
- A few purple loosestrife occurrences were mapped along the northern shore in 2021

Planning Meeting II

Primary Objective: Create implementation plan framework

Steps to Achieve Objective:

1. Discuss challenges facing the lake and the lake group
2. Convert challenges to management goals
3. Create management actions to meet management goals
4. Determine timeframes and facilitators to carry out actions

Assignment for Planning Meeting II

1. Create list of challenges facing lake and lake group – keep for meeting
2. Review stakeholder survey results
3. Send potential report section edits and questions to Todd

B

APPENDIX B

Stakeholder Survey Response Charts and Comments

Silver Lake - Anonymous Stakeholder Survey

Surveys Distributed: 27
Surveys Returned: 13
Response Rate: 48%

Silver Lake Property

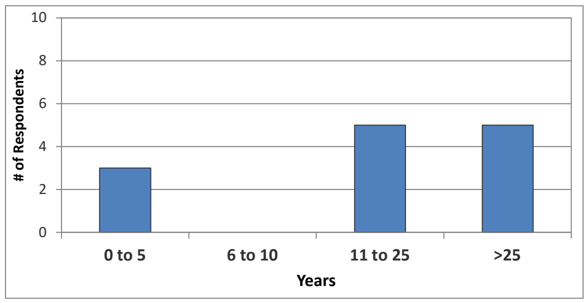
1. Is your property on the lake or off the lake? Please select one choice.

Answer Options	Response Percent	Response Count
On the lake	100.0%	13
Off the lake	0.0%	0
	answered question	**13**
	skipped question	**0**

2. How many years have you owned or rented your property on or near Silver Lake?

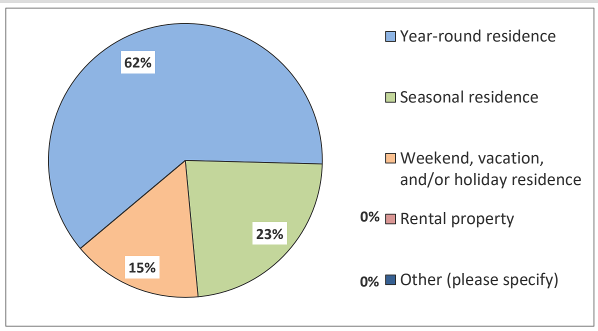
Answer Options	Response Count
	13
answered question	**13**
skipped question	**0**

Category (# of years)	Responses	% Response
0 to 5	**3**	**23%**
6 to 10	**0**	**0%**
11 to 25	**5**	**38%**
>25	**5**	**38%**



3. How is your property on or near Silver Lake used?

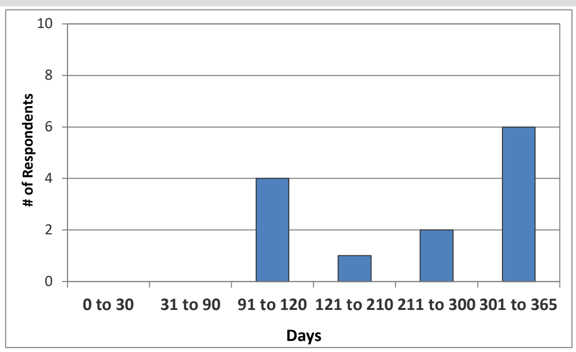
Answer Options	Response Percent	Response Count
Year-round residence	61.5%	8
Seasonal residence	23.1%	3
Weekend, vacation, and/or holiday residence	15.4%	2
Rental property	0.0%	0
Other (please specify)	0.0%	0
	answered question	**13**
	skipped question	**0**



4. Considering the past three years, how many days each year is your property used by you or others?

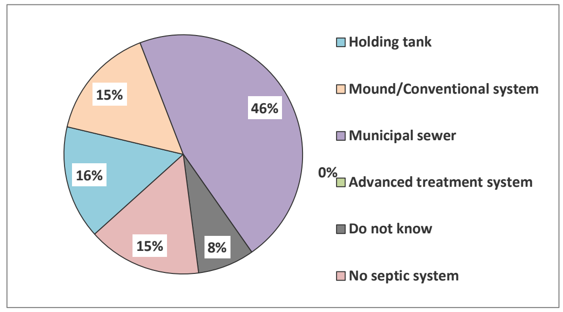
Answer Options	Response Count
	13
answered question	**13**
skipped question	**0**

Category (# of days)	Responses	%
0 to 30	**0**	**0%**
31 to 90	**0**	**0%**
91 to 120	**4**	**31%**
121 to 210	**1**	**8%**
211 to 300	**2**	**15%**
301 to 365	**6**	**46%**



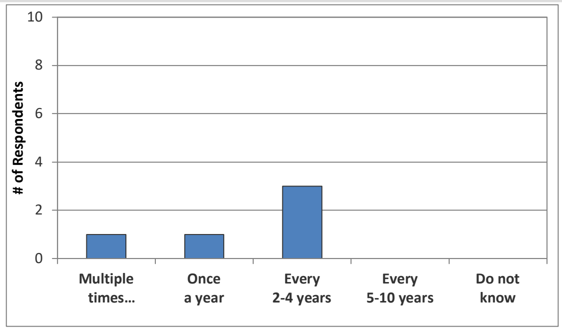
5. What type of septic system does your property have?

Answer Options	Response Percent	Response Count
Holding tank	15.4%	2
Mound/Conventional system	15.4%	2
Municipal sewer	46.2%	6
Advanced treatment system	0.0%	0
Do not know	7.7%	1
No septic system	15.4%	2
	answered question	**13**
	skipped question	**0**



6. How often is the septic system on your property pumped?

Answer Options	Response Percent	Response Count
Multiple times a year	20.0%	1
Once a year	20.0%	1
Every 2-4 years	60.0%	3
Every 5-10 years	0.0%	0
Do not know	0.0%	0
	answered question	**5**
	skipped question	**8**

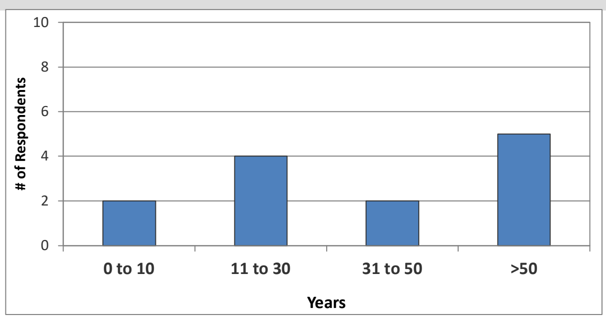


Recreational Activity on Silver Lake

7. How many years ago did you first visit Silver Lake?

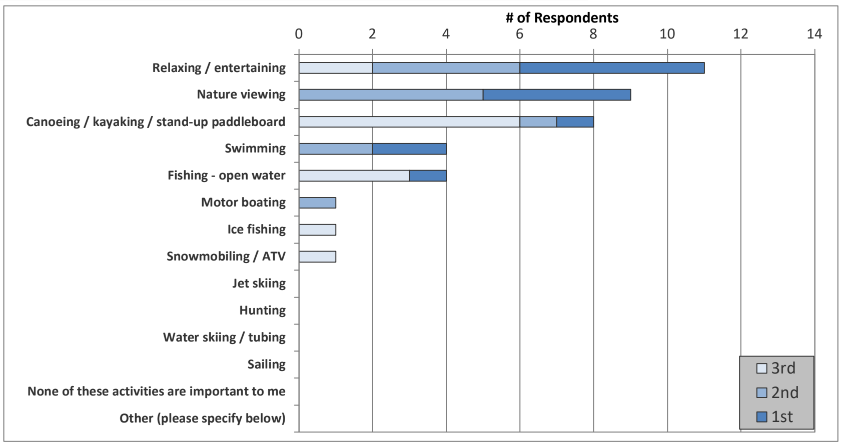
Answer Options	Response Count
	13
answered question	**13**
skipped question	**0**

Category (# of years)	**Responses**	**% Response**
0 to 10	2	**15%**
11 to 30	4	**31%**
31 to 50	2	**15%**
>50	5	**38%**



8. Please rank up to three activities that are important reasons for owning your property on or near Silver Lake, with 1 being the most important.

Answer Options	1st	2nd	3rd	Weighted Average	Response Count
Relaxing / entertaining	5	4	2	1.73	11
Nature viewing	4	5	0	1.56	9
Canoeing / kayaking / stand-up paddleboard	1	1	6	2.63	8
Swimming	2	2	0	1.5	4
Fishing - open water	1	0	3	2.5	4
Motor boating	0	1	0	2	1
Ice fishing	0	0	1	3	1
Snowmobiling / ATV	0	0	1	3	1
Jet skiing	0	0	0	0	0
Hunting	0	0	0	0	0
Water skiing / tubing	0	0	0	0	0
Sailing	0	0	0	0	0
None of these activities are important to me	0	0	0	0	0
Other (please specify below)	0	0	0	0	0
				answered question	**13**
				skipped question	**0**

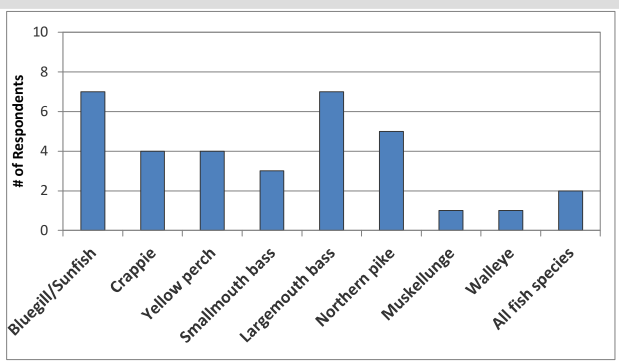


9. Have you personally fished on Silver Lake in the past three years?

Answer Options	Response Percent	Response Count
Yes	69.2%	9
No	30.8%	4
	answered question	**13**
	skipped question	**0**

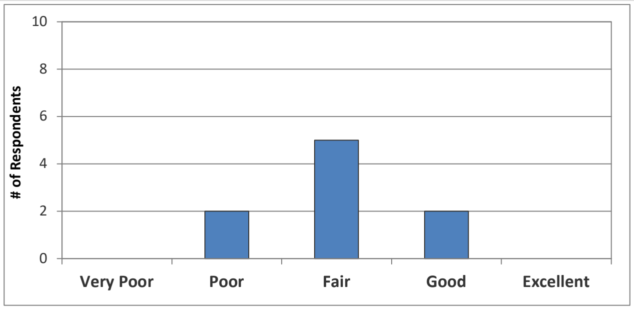
10. What species of fish do you try to catch on Silver Lake?

Answer Options	Response Percent	Response Count
Bluegill/Sunfish	77.8%	7
Crappie	44.4%	4
Yellow perch	44.4%	4
Smallmouth bass	33.3%	3
Largemouth bass	77.8%	7
Northern pike	55.6%	5
Muskellunge	11.1%	1
Walleye	11.1%	1
All fish species	22.2%	2
Other	0.0%	0
	answered question	**9**
	skipped question	**4**



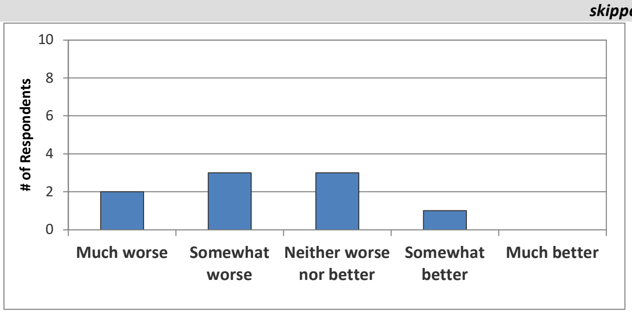
11. How would you describe the current quality of fishing on Silver Lake?

Answer Options	Very Poor	Poor	Fair	Good	Excellent	Response Count
	0	2	5	2	0	9
					answered question	9
					skipped question	4



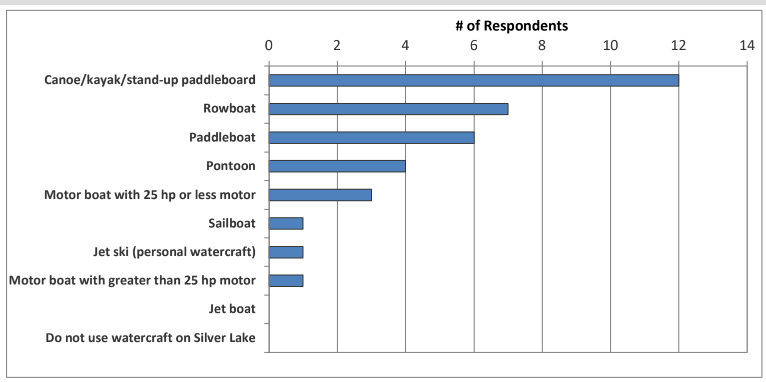
12. How has the quality of fishing changed on Silver Lake since you have started fishing the lake?

Answer Options	Much worse	Somewhat worse	Neither worse nor better	Somewhat better	Much better	Response Count
	2	3	3	1	0	9
					answered question	9
					skipped question	4



13. What types of watercraft do you currently use on Silver Lake?

Answer Options	Response Percent	Response Count
Canoe/kayak/stand-up paddleboard	92.3%	12
Rowboat	53.9%	7
Paddleboat	46.2%	6
Pontoon	30.8%	4
Motor boat with 25 hp or less motor	23.1%	3
Sailboat	7.7%	1
Jet ski (personal watercraft)	7.7%	1
Motor boat with greater than 25 hp motor	7.7%	1
Jet boat	0.0%	0
Do not use watercraft on Silver Lake	0.0%	0
	answered question	13
	skipped question	0



14. Do you use your watercraft on waters other than Silver Lake?

Answer Options	Response Percent	Response Count
Yes	23.1%	3
No	76.9%	10
	answered question	13
	skipped question	0

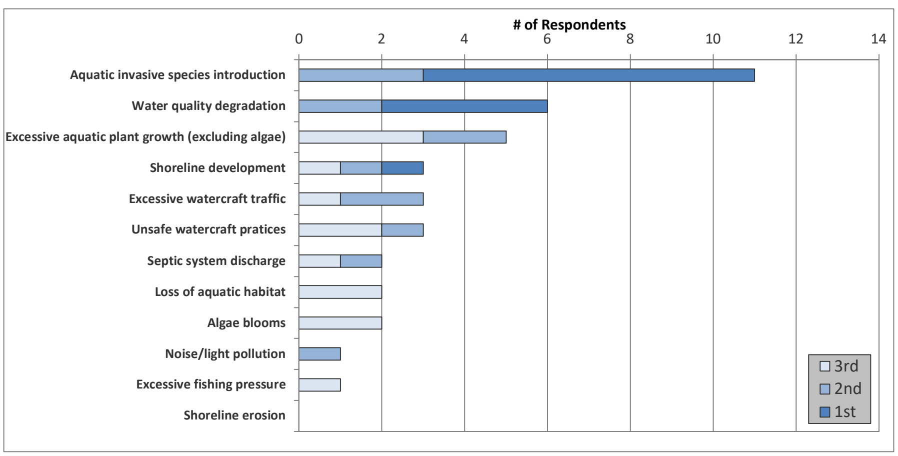
15. What is your typical cleaning routine after using your watercraft on waters other than Silver Lake?

Answer Options	Response Percent	Response Count
Remove aquatic hitchhikers (ex. plant material, clams, mussels)	100.0%	4
Drain bilge	50.0%	2
Rinse boat	75.0%	3
Power wash boat	25.0%	1
Apply bleach	0.0%	0
Air dry boat for 5 or more days	75.0%	3
Do not clean boat	0.0%	0
Other		0
	answered question	4
	skipped question	9

Silver Lake Current and Historic Condition, Health and Management

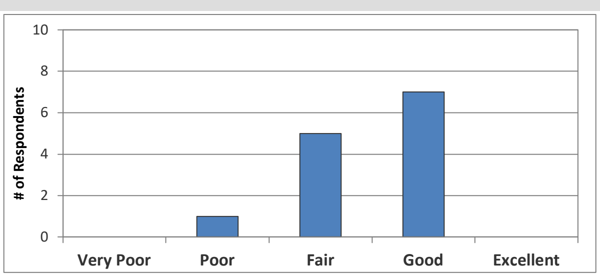
16. From the list below, please rank your top three concerns regarding Silver Lake, with 1 being your greatest concern.

Answer Options	1st	2nd	3rd	Response Count
Aquatic invasive species introduction	8	3	0	11
Water quality degradation	4	2	0	6
Excessive aquatic plant growth (excluding algae)	0	2	3	5
Shoreline development	1	1	1	3
Excessive watercraft traffic	0	2	1	3
Unsafe watercraft pratices	0	1	2	3
Septic system discharge	0	1	1	2
Loss of aquatic habitat	0	0	2	2
Algae blooms	0	0	2	2
Noise/light pollution	0	1	0	1
Excessive fishing pressure	0	0	1	1
Shoreline erosion	0	0	0	0
Other (please specify)	0	0	0	0
			answered question	13
			skipped question	0



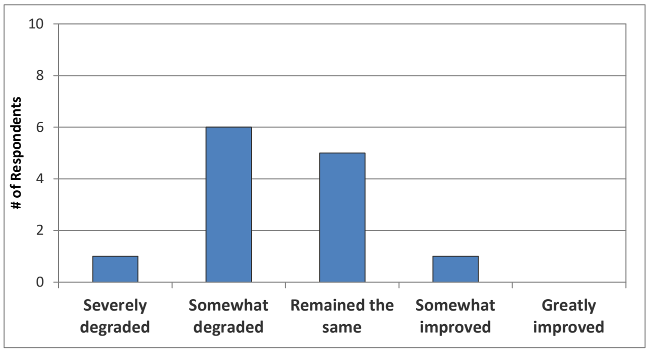
17. How would you describe the overall current water quality of Silver Lake?

Answer Options	Very Poor	Poor	Fair	Good	Excellent	Response Count
	0	1	5	7	0	13
					answered question	13
					skipped question	0



18. How has the overall water quality changed in Silver Lake since you first visited the lake?

Answer Options	Severely degraded	Somewhat degraded	Remained the same	Somewhat improved	Greatly improved	Response Count
	1	6	5	1	0	13
					answered question	**13**
					skipped question	**0**



19. Which of the following would you say is the single most important aspect when considering water quality?

Answer Options	Response Percent	Response Count
Aquatic plant growth (not including algae blooms)	46.2%	6
Water clarity (clearness of water)	30.8%	4
Algae blooms	15.4%	2
Smell/odors	7.7%	1
Water color	0.0%	0
Water level	0.0%	0
Fish kills	0.0%	0
Other	0.0%	0
	answered question	**13**
	skipped question	**0**

20. Using the following scale, what impact, if any, do you believe each of the following practices have on the water quality of Silver Lake?

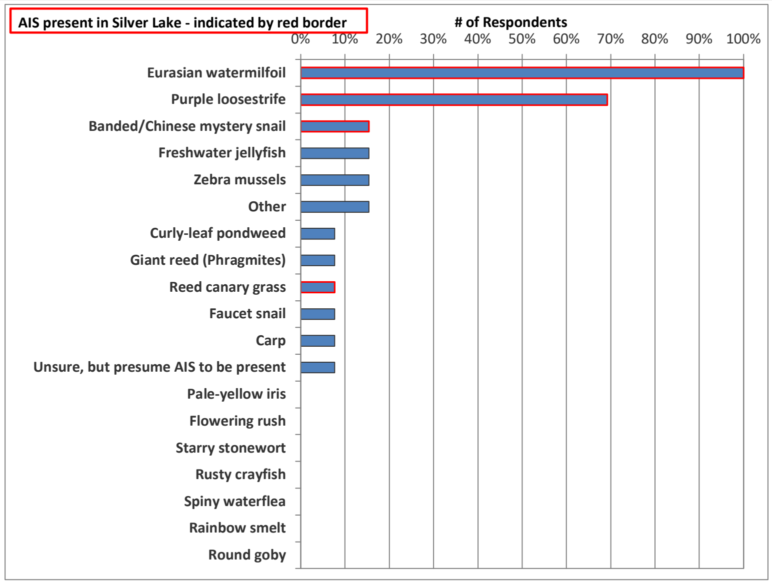
Answer Options	Large negative impact	Small negative impact	No impact	Small positive impact	Large positive impact	Unsure/ Need more info.	Response Count	Weighted Average
Failing septic systems	4	3	1	0	0	4	12	1.08
Runoff from impervious surfaces, such as concrete	4	5	3	0	0	1	13	1.77
Installation of sand or pea gravel swimming	0	2	7	1	1	2	13	2.62
Large-scale removal of native aquatic plants	3	2	3	3	1	1	13	2.54
Large-scale removal of invasive aquatic plants	1	1	1	1	8	1	13	3.85
Operation of watercraft at wake speeds in shallow water areas	2	3	4	0	2	2	13	2.31
Rain gutters and downspouts draining toward the lake	0	3	8	1	0	1	13	2.62
Removal of near-shore emergent vegetation, such as bulrushes, lily pads, cattails, etc.	1	7	3	1	0	1	13	2.15
Removal of upland vegetation in shoreline buffer areas	2	5	2	2	0	2	13	2
Removal of shoreline woody debris in the lake, such as downed trees	1	4	1	2	2	3	13	2.31
Shoreline alterations (rip-rap retaining walls, etc.)	2	2	5	2	0	2	13	2.23
						answered question	**13**	
						skipped question	**0**	

21. Before reading the statement provided, had you ever heard of aquatic invasive species?		
Answer Options	**Response Percent**	**Response Count**
Yes	100.0%	13
No	0.0%	0
	answered question	**13**
	skipped question	**0**

22. Do you believe aquatic invasive species are present within Silver Lake?		
Answer Options	**Response Percent**	**Response Count**
Yes	100.0%	13
No	0.0%	0
	answered question	**13**
	skipped question	**0**

23. Which aquatic invasive species do you believe are present in or immediately around Silver Lake?

Answer Options	**Response Percent**	**Response Count**
Eurasian watermilfoil	100.0%	13
Purple loosestrife	69.2%	9
Banded/Chinese mystery snail	15.4%	2
Freshwater jellyfish	15.4%	2
Zebra mussels	15.4%	2
Other	15.4%	2
Curly-leaf pondweed	7.7%	1
Giant reed (Phragmites)	7.7%	1
Reed canary grass	7.7%	1
Faucet snail	7.7%	1
Carp	7.7%	1
Unsure, but presume AIS to be present	7.7%	1
Pale-yellow iris	0.0%	0
Flowering rush	0.0%	0
Starry stonewort	0.0%	0
Rusty crayfish	0.0%	0
Spiny waterflea	0.0%	0
Rainbow smelt	0.0%	0
Round goby	0.0%	0
	answered question	**13**
	skipped question	**0**



Number	**"Other" responses**
1	Japanese knotweed
2	

24. Before the present year, aquatic herbicides have been used to manage Eurasian watermilfoil on Silver Lake. Professional monitoring of the aquatic plant community has also occurred during this time. Prior to reading this information, did you know that aquatic herbicides were being applied in Silver Lake to manage Eurasian watermilfoil?

Answer Options	**Response Percent**	**Response Count**
Yes	92.3%	12
I think so but can't say for certain	7.7%	1
No	0.0%	0
	answered question	**13**
	skipped question	**0**

25. What is your level of support or opposition for the <u>past</u> use of aquatic herbicides to treat Eurasian watermilfoil in Silver Lake in previous years?

Answer Options	**Completely oppose**	**Moderately oppose**	**Neither oppose nor support**	**Moderately support**	**Completely support**		**Response Count**
	1	0	3	2	7		13
						answered question	**13**
						skipped question	

26. What is your level of support or opposition for the future use of aquatic herbicides to target Eurasian watermilfoil in Silver Lake?

Answer Options	**Completely oppose**	**Moderately oppose**	**Neither oppose nor support**	**Moderately support**	**Completely support**		**Response Count**
	1	0	0	6	6		13
						answered question	**13**
						skipped question	**0**

27. If you selected "Moderately oppose" or "Completely oppose" for Question #26, what is the reason or reasons you oppose the future use of aquatic herbicides to target Eurasian watermilfoil in Silver Lake?

Answer Options	Response Percent	Response Count
Potential cost of treatment is too high	0.0%	0
Potential impacts to native aquatic plant species	100.0%	1
Potential impacts to native (non-plant) species (fish, insects, etc.)	100.0%	1
Potential impacts to human health	100.0%	1
Future impacts are unknown	100.0%	1
Ineffectiveness of herbicide strategy	100.0%	1
Another reason	0.0%	0
answered question		**1**
skipped question		**12**

Note: Only one response in Question 26 was answered as "completely oppose" so the answers here are from that one respondent.

28. Has the Eurasian watermilfoil population ever had a negative impact on your enjoyment of Silver Lake?

	Yes		Unsure		No		
Answer Options	**Response Percent**	**Response Count**	**Response Percent**	**Response Count**	**Response Percent**	**Response Count**	Total
Aesthetics	53.9%	7	23.1%	3	23.1%	3	13
Swimming	53.9%	7	7.7%	1	38.5%	5	13
Canoeing/kayaking/stand-up paddleboard	50.0%	6	8.3%	1	41.7%	5	12
Motor boating	46.2%	6	15.4%	2	38.5%	5	13
Fishing – Open water	38.5%	5	23.1%	3	38.5%	5	13
Nature Viewing	23.1%	3	30.8%	4	46.2%	6	13
Ice fishing	16.7%	2	25.0%	3	58.3%	7	12
Other (please specify)	0.0%	0	100.0%	1	0.0%	0	1
answered question	**13**						
skipped question	**0**						

Eagle River Silver Lake Association

29. Before receiving this mailing, had you ever heard of the Silver Lake Association?

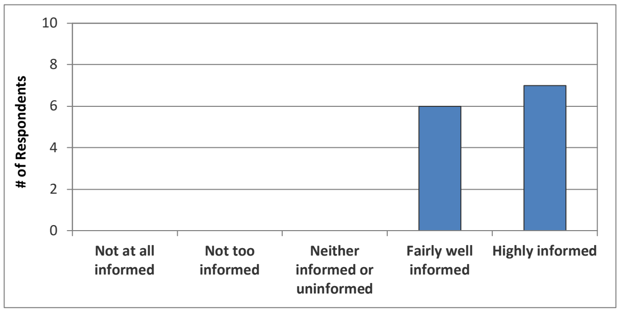
Answer Options	Response Percent	Response Count
Yes	100.0%	13
No	0.0%	0
answered question		**13**
skipped question		**0**

30. What is your membership status with the Silver Lake Association?

Answer Options	Response Percent	Response Count
Current member	100.0%	13
Former member	0.0%	0
Never been a member	0.0%	0
answered question		**13**
skipped question		**0**

31. How informed has (or had) the Silver Lake Association kept you regarding issues with Silver Lake and its management?

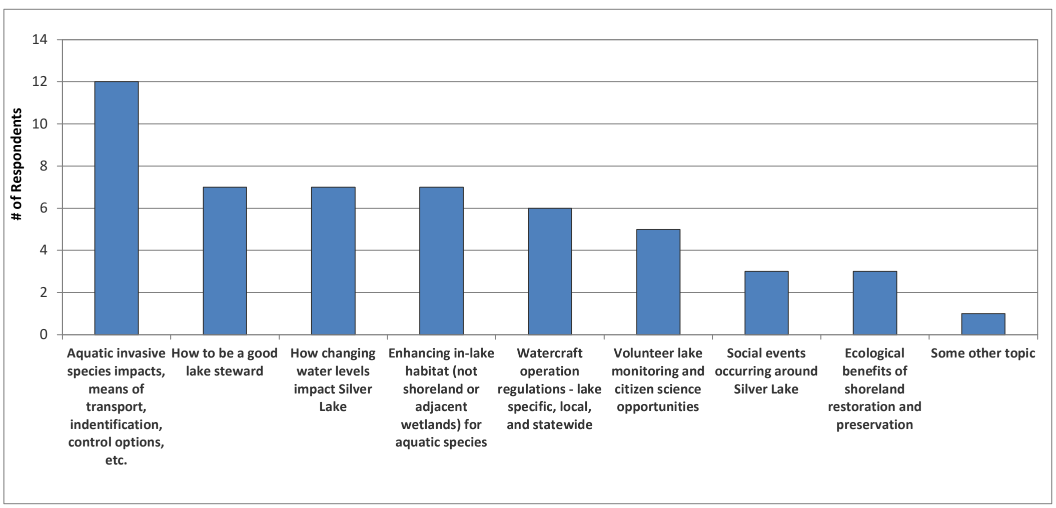
Answer Options	Not at all informed	Not too informed	Neither informed or uninformed	Fairly well informed	Highly informed	Response Count
	0	0	0	6	7	13
					answered question	**13**
					skipped question	**0**



32. Stakeholder education is an important component of every lake management planning effort. Which of these subjects would you like to learn more about?

Answer Options	Response Percent	Response Count
Aquatic invasive species impacts, means of transport, indentification, control options, etc.	92.3%	12
How to be a good lake steward	53.9%	7
How changing water levels impact Silver Lake	53.9%	7
Enhancing in-lake habitat (not shoreland or adjacent wetlands) for aquatic species	53.9%	7
Watercraft operation regulations - lake specific, local, and statewide	46.2%	6
Volunteer lake monitoring and citizen science opportunities	38.5%	5
Social events occurring around Silver Lake	23.1%	3
Ecological benefits of shoreland restoration and preservation	23.1%	3
Some other topic	7.7%	1
Not interested in learning more on any of these subjects	0.0%	0
	answered question	**13**
	skipped question	**0**

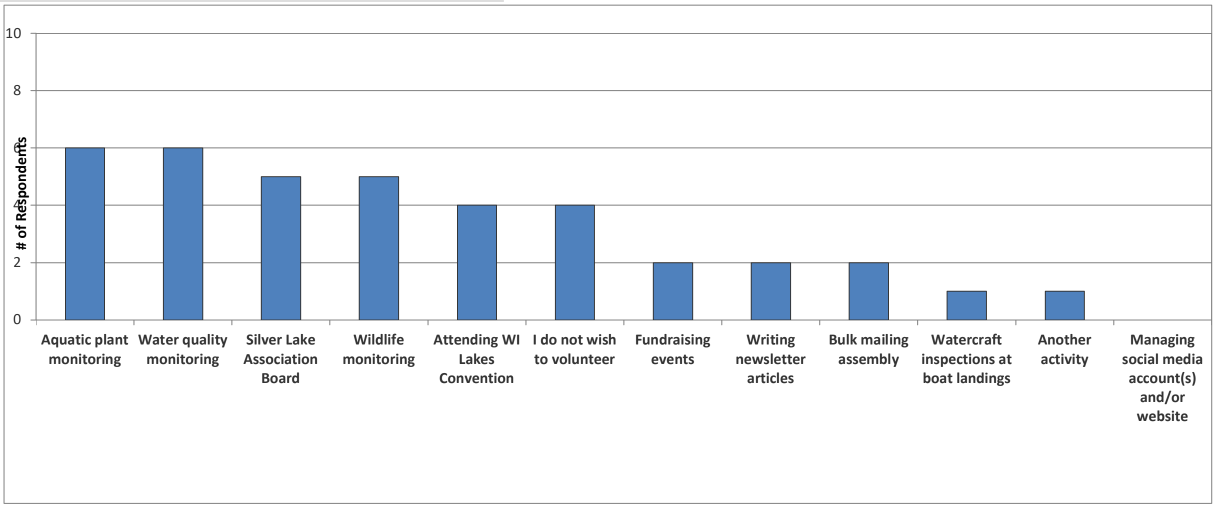
Number	Other (please specify)
1	Bioremediation of AIS (e.g. milfoil weevil)
2	



33. The effective management of Silver Lake will require the cooperative efforts of numerous volunteers. Please circle the activities you would be willing to participate in if additional assistance is required.

Answer Options	Response Percent	Response Count
Aquatic plant monitoring	46.2%	6
Water quality monitoring	46.2%	6
Silver Lake Association Board	38.5%	5
Wildlife monitoring	38.5%	5
Attending WI Lakes Convention	30.8%	4
I do not wish to volunteer	30.8%	4
Fundraising events	15.4%	2
Writing newsletter articles	15.4%	2
Bulk mailing assembly	15.4%	2
Watercraft inspections at boat landings	7.7%	1
Another activity	7.7%	1
Managing social media account(s) and/or wek	0.0%	0
answered question		13
skipped question		0

Number	Another Activity (please specify)
1	Raising weevils for bioremediation of EWM
2	
3	



34. Please feel free to provide written comments concerning Silver Lake, its current and/or historic condition and its management.

Answer Options	Response Count
	4
answered question	4
skipped question	9

Number	Response Text
1	I am a fisherman and want the invasive milfoil controlled if possible
2	Major concern with shoreline development, not sure how/ Why Silver Lake is exempt from State DNR rules? Set back code for the State is 75 feet but on Silver Lake you can build a house, garage etc 25 feet from the shoreline. Makes no sense to me
3	Since Silver Lake is host to many entities who don't necessarily live on it, i would like to see the concepts of "stakeholder" and "stewardship" involve this larger population. (e.g. when vegetation encroaches on the area cut for the ice castle, thereby making rotten blocks that melt quickly, the city is a stakeholder.) I feel it would be to everyone's advantage to involve, say, Trees For Tomorrow and the Northland Pines district as stakeholders as well.
4	I believe that, in addition to recent levels of rainfall, a somewhat restricted culvert has resulted in higher than normal water levels in Silver Lake.

C

APPENDIX C

Water Quality Data

Silver Lake

	Secchi (feet)				Chlorophyll-*a* (µg/L)				Total Phosphorus (µg/L)			
	Growing Season		Summer		Growing Season		Summer		Growing Season		Summer	
Year	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**
1979	1	11.5	1	11.5	1	8.2	1	8.2				
1993	1	7.5	1	7.5								
1994	6	10.0	4	9.2								
1995	3	11.6	1	9.0								
1996	2	8.0	0									
1997	1	10.0	0									
1998	1	8.0	0									
1999	0		0									
2000	0		0									
2001	0		0									
2002	0		0									
2003	0		0									
2004	0		0									
2005	0		0									
2006	0		0									
2007	0		0									
2008	0		0									
2009	0		0									
2010	1	9.3	0									
2011	0		0									
2012	0		0									
2013	0		0									
2014	0		0									
2015	0		0		10	3.0	9	2.9				
2016	0		0		10	3.5	8	3.2				
2017	0		0		9	5.0	8	4.7				
2018	0		0		0		0					
2019	0		0		0		0					
2020	0		0		0		0					
2021	5	10.2	3	12.5	5	3.8	3	2.2	5	15.5	3.0	15.2
All Years (Weighted)		9.9		10.2		3.9		3.6		15.5		15.2
DHDL Median				10.8				5.0				17.0
NLF Ecoregion Median				8.9				5.6				21.0

D

APPENDIX D

Point-Intercept Aquatic Macrophyte Survey Data

Silver Lake

	Scientific Name	Common Name	LFOO (%) 2005	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Dicots	*Myriophyllum tenellum*	Dwarf watermilfoil	14.4	19.7	15.4	23.0	19.3	19.2	17.7	22.0	20.2	15.9	13.8	11.8	21.0	8.2	9.9	16.0
	Myriophyllum spicatum	Eurasian watermilfoil	6.1	0.0	0.0	0.0	0.0	0.0	0.0	0.8	3.9	3.0	0.0	0.0	2.5	1.8	2.8	9.2
	Utricularia resupinata	Northeastern bladderwort	0.0	0.0	0.8	0.7	2.2	1.5	2.3	2.4	0.8	0.0	0.0	0.0	6.7	3.6	2.8	1.7
	Nuphar variegata	Spatterdock	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0
	Utricularia cornuta	Horned bladderwort	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ceratophyllum demersum	Coontail	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Non-dicots	*Vallisneria americana*	Wild celery	21.2	36.2	32.3	37.0	42.2	50.0	49.2	55.1	55.0	54.5	52.9	63.2	73.9	84.5	59.6	71.4
	Chara & Nitella	Charophytes	40.2	41.7	50.8	39.3	52.6	48.5	53.8	58.3	40.3	62.9	73.2	66.2	36.1	56.4	48.9	63.0
	Elodea canadensis	Common waterweed	38.6	47.2	53.1	38.5	48.9	49.2	46.9	34.6	40.3	35.6	21.0	32.4	38.7	38.2	33.3	43.7
	Potamogeton amplifolius	Large-leaf pondweed	30.3	31.5	33.8	38.5	31.9	29.2	33.8	31.5	31.8	22.0	29.0	25.7	38.7	40.0	31.9	46.2
	Chara spp.	Muskgrasses	0.0	15.7	15.4	16.3	22.2	23.8	19.2	57.5	27.1	59.1	65.2	60.3	36.1	56.4	9.9	0.0
	Nitella spp.	Stoneworts	40.2	32.3	40.8	28.9	30.4	30.8	41.5	1.6	16.3	4.5	8.7	8.8	0.0	0.0	12.8	4.2
	Eleocharis acicularis	Needle spikerush	0.0	0.0	3.1	13.3	14.8	20.0	18.5	22.0	24.8	19.7	0.0	13.2	2.5	20.9	17.0	28.6
	Potamogeton pusillus	Small pondweed	5.3	0.8	3.1	3.0	8.1	18.5	18.5	15.0	20.2	15.2	13.8	2.2	2.5	1.8	1.4	5.0
	Potamogeton epihydrus	Ribbon-leaf pondweed	0.0	0.8	6.2	9.6	10.4	12.3	21.5	11.8	5.4	3.0	8.0	1.5	2.5	2.7	11.3	5.9
	Najas flexilis	Slender naiad	2.3	1.6	16.2	7.4	7.4	16.2	4.6	4.7	3.1	4.5	2.2	0.0	2.5	6.4	3.5	8.4
	Fissidens spp. & Fontinalis spp.	Aquatic Moss	0.0	1.6	4.6	4.4	1.5	1.5	6.9	4.7	2.3	3.0	7.2	5.1	0.0	15.5	4.3	12.6
	Potamogeton vaseyi	Vasey's pondweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	7.6	0.0	12.8	15.1
	Potamogeton diversifolius	Water-thread pondweed	0.0	0.0	0.0	0.0	1.5	3.8	6.9	13.4	14.0	2.3	0.0	0.0	0.0	2.7	0.7	1.7
	Isoetes spp.	Quillwort spp.	0.8	2.4	4.6	6.7	3.7	3.8	3.1	3.1	0.0	0.8	0.0	1.5	4.2	4.5	2.1	0.0
	Eleocharis robbinsii	Robbins' spikerush	0.0	0.0	8.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	24.4	0.0	0.0	0.0
	Lobelia dortmanna	Water lobelia	6.8	0.0	0.8	0.0	0.0	0.8	1.5	0.8	1.6	0.8	1.4	0.7	4.2	2.8	1.4	0.8
	Sparganium angustifolium	Narrow-leaf bur-reed	0.0	2.4	0.8	0.0	0.0	2.3	3.1	2.4	2.3	2.3	0.0	0.0	0.0	0.0	0.0	0.0
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	5.9
	Potamogeton spirillus	Spiral-fruited pondweed	3.8	0.0	3.1	0.0	0.0	1.5	0.0	0.8	0.0	0.0	0.0	0.0	0.0	3.6	0.0	0.8
	Elodea nuttallii	Slender waterweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.0	0.0	0.0	0.0	0.0	0.0
	Eleocharis palustris	Creeping spikerush	3.0	0.0	0.8	0.0	0.7	0.8	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0
	Juncus pelocarpus	Brown-fruited rush	0.0	0.0	0.0	0.0	0.0	0.8	0.8	0.8	0.8	0.0	0.0	0.0	1.7	0.0	0.7	0.0
	Sparganium eurycarpum	Common bur-reed	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sagittaria rigida	Stiff arrowhead	1.5	0.0	1.5	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Stuckenia pectinata	Sago pondweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0
	Potamogeton foliosus	Leafy pondweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	0.0
	Lemna trisulca	Forked duckweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.0	0.0
	Elatine minima	Waterwort	0.0	0.8	0.8	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sparganium sp.	Bur-reed sp.	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sagittaria graminea	Grass-leaved arrowhead	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.0
	Schoenoplectus subterminalis	Water bulrush	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Potamogeton gramineus	Variable-leaf pondweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0
	Heteranthera dubia	Water stargrass	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0
	Eriocaulon aquaticum	Pipewort	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Silver Lake

	Scientific Name	Common Name	2005-2007 % Change	2005-2007 Direction	2007-2008 % Change	2007-2008 Direction	2008-2009 % Change	2008-2009 Direction	2009-2010 % Change	2009-2010 Direction	2010-2011 % Change	2010-2011 Direction	2011-2012 % Change	2011-2012 Direction	2012-2013 % Change	2012-2013 Direction	2013-2014 % Change	2013-2014 Direction
Dicots	*Myriophyllum tenellum*	Dwarf watermilfoil	36.8	▲	-21.8	▼	49.3	▲	-16.1	▼	-0.1	▼	-8.0	▼	24.6	▲	-8.6	▼
	Myriophyllum spicatum	Eurasian watermilfoil	-100.0	**▼**		-		-		-		-		-		▲	392.2	▲
	Utricularia resupinata	Northeastern bladderwort		-		▲	-3.7	▼	200.0	▲	-30.8	▼	50.0	▲	2.4	▲	-67.2	▼
	Nuphar variegata	Spatterdock		▲	-100.0	▼		-		-		-		-		▲	-100.0	▼
	Utricularia cornuta	Horned bladderwort		▲	-100.0	▼		-		-		-		-		-		-
	Ceratophyllum demersum	Coontail		-		▲	-100.0	▼		-		-		-		-		-
Non-dicots	*Vallisneria americana*	Wild celery	70.8	**▲**	-10.8	▼	14.6	▲	14.0	▲	18.4	▲	-1.5	▼	12.0	▲	-0.1	▼
	Chara & Nitella	Charophytes	3.9	▲	21.7	▲	-22.7	▼	34.0	**▲**	-7.9	▼	11.1	▲	8.2	▲	-30.8	**▼**
	Elodea canadensis	Common waterweed	22.3	▲	12.3	▲	-27.4	**▼**	26.9	▲	0.7	▲	-4.7	▼	-26.2	**▼**	16.3	▲
	Potamogeton amplifolius	Large-leaf pondweed	3.9	▲	7.5	▲	13.8	▲	-17.3	▼	-8.2	▼	15.8	▲	-6.9	▼	0.9	▲
	Chara spp.	Muskgrasses		**▲**	-2.3	▼	5.9	▲	36.4	▲	7.3	▲	-19.4	▼	198.9	**▲**	-52.8	**▼**
	Nitella spp.	Stoneworts	-19.6	▼	26.3	▲	-29.1	**▼**	5.1	▲	1.3	▲	35.0	▲	-96.2	**▼**	933.7	**▲**
	Eleocharis acicularis	Needle spikerush		-		**▲**	333.3	**▲**	11.1	▲	35.0	▲	-7.7	▼	19.4	▲	12.5	▲
	Potamogeton pusillus	Small pondweed	-85.2	**▼**	290.8	▲	-3.7	▼	175.0	▲	126.6	**▲**	0.0	-	-19.0	▼	34.7	▲
	Potamogeton epihydrus	Ribbon-leaf pondweed		▲	681.5	**▲**	56.5	▲	7.7	▲	18.7	▲	75.0	**▲**	-45.2	**▼**	-54.1	▼
	Najas flexilis	Slender naiad	-30.7	▼	925.8	**▲**	-54.1	**▼**	0.0	-	118.1	**▲**	-71.4	**▼**	2.4	▲	-34.4	▼
	Fissidens spp. & Fontinalis spp.	Aquatic Moss		▲	193.1	▲	-3.7	▼	-66.7	▼	3.8	▲	350.0	**▲**	-31.8	▼	-50.8	▼
	Potamogeton vaseyi	Vasey's pondweed		-		-		-		-		-		-		-		-
	Potamogeton diversifolius	Water-thread pondweed		-		-		-		▲	159.6	▲	80.0	▲	93.4	▲	4.2	▲
	Isoetes spp.	Quillwort spp.	211.8	▲	95.4	▲	44.4	▲	-44.4	▼	3.8	▲	-20.0	▼	2.4	▲	-100.0	**▼**
	Eleocharis robbinsii	Robbins' spikerush		-		**▲**	-100.0	**▼**		-		-		-		-		-
	Lobelia dortmanna	Water lobelia	-100.0	**▼**		▲	-100.0	▼		-		▲	100.0	▲	-48.8	▼	96.9	▲
	Sparganium angustifolium	Narrow-leaf bur-reed		▲	-67.4	▼	-100.0	▼		-		▲	33.3	▲	-23.2	▼	-1.6	▼
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)		-		-		▲	-100.0	▼		-		-		-		-
	Potamogeton spirillus	Spiral-fruited pondweed	-100.0	**▼**		**▲**	-100.0	**▼**		-		▲	-100.0	▼		▲	-100.0	▼
	Elodea nuttallii	Slender waterweed		-		-		-		-		-		-		-		-
	Eleocharis palustris	Creeping spikerush	-100.0	**▼**		▲	-100.0	▼		▲	3.8	▲	0.0	-	-100.0	▼		-
	Juncus pelocarpus	Brown-fruited rush		-		-		-		-		▲	0.0	-	2.4	▲	-1.6	▼
	Sparganium eurycarpum	Common bur-reed	-100.0	**▼**		-		-		-		-		-		-		-
	Sagittaria rigida	Stiff arrowhead	-100.0	▼		▲	-100.0	▼		-		-		-		▲	-100.0	▼
	Stuckenia pectinata	Sago pondweed		-		-		-		-		-		-		-		-
	Potamogeton foliosus	Leafy pondweed		-		-		-		-		-		-		-		-
	Lemna trisulca	Forked duckweed		-		-		-		-		-		-		-		-
	Elatine minima	Waterwort		▲	-2.3	▼	-100.0	▼		-		▲	-100.0	▼		-		-
	Sparganium sp.	Bur-reed sp.		-		-		-		▲	-100.0	▼		-		-		-
	Sagittaria graminea	Grass-leaved arrowhead		-		-		-		-		-		-		-		-
	Schoenoplectus subterminalis	Water bulrush	-100.0	▼		-		-		-		-		-		-		-
	Potamogeton gramineus	Variable-leaf pondweed		-		-		-		-		-		-		-		-
	Heteranthera dubia	Water stargrass		-		-		-		-		-		-		-		-
	Eriocaulon aquaticum	Pipewort		▲	-100.0	▼		-		-		-		-		-		-

▲ or **▼** = Change Statistically Valid (Chi-square; α = 0.05)
▲ or ▼ = Change Not Statistically Valid (Chi-square; α = 0.05)

Silver Lake

	Scientific Name	Common Name	2014-2015		2015-2016		2016-2017		2017-2018		2018-2019		2019-2020		2020-2021	
			% Change	Direction	% Change	Direction	% Change	Direction	% Change	Direction	% Change	Direction	% Change	Direction	% Change	Direction
Dicots	*Myriophyllum tenellum*	Dwarf watermilfoil	-21.1	▽	-13.5	▽	-14.6	▽	78.6	▲	-61.1	▼	21.4	△	60.8	△
	Myriophyllum spicatum	Eurasian watermilfoil	-21.8	▽	-100.0	▼		-		△	-27.9	▽	56.0	△	225.8	▲
	Utricularia resupinata	Northeastern bladderwort	-100.0	▽		-		-		▲	-45.9	▽	-22.0	▽	-40.8	▽
	Nuphar variegata	Spatterdock		-		-		-		-		△	-100.0	▽		-
	Utricularia cornuta	Horned bladderwort		-		-		-		-		-		-		-
	Ceratophyllum demersum	Coontail		-		-		-		-		-		-		-
Non-dicots	*Vallisneria americana*	Wild celery	-0.9	▽	-3.0	▽	19.5	△	16.9	△	14.3	▲	-29.5	▼	19.9	▲
	Chara & Nitella	Charophytes	56.0	▲	16.4	△	-9.6	▽	-45.4	▼	56.0	▲	-13.2	▽	28.8	▲
	Elodea canadensis	Common waterweed	-11.7	▽	-41.0	▼	54.0	▲	19.5	△	-1.2	▽	-12.7	▽	31.1	△
	Potamogeton amplifolius	Large-leaf pondweed	-30.9	▽	31.9	△	-11.2	▽	50.2	▲	3.5	△	-20.2	▽	44.8	▲
	Chara spp.	Muskgrasses	117.8	▲	10.4	△	-7.5	▽	-40.1	▼	56.0	▲	-82.4	▼	-100.0	▼
	Nitella spp.	Stoneworts	-72.1	▼	91.3	△	1.5	△	-100.0	▼		-		▲	-67.1	▼
	Eleocharis acicularis	Needle spikerush	-20.6	▽	-100.0	▼		▲	-81.0	▼	729.4	▲	-18.6	▽	67.9	▲
	Potamogeton pusillus	Small pondweed	-24.8	▽	-9.1	▽	-84.0	▼	14.3	△	-27.9	▽	-22.0	▽	255.5	△
	Potamogeton epihydrus	Ribbon-leaf pondweed	-44.2	▽	163.0	△	-81.6	▼	71.4	△	8.2	△	316.1	▲	-48.2	▽
	Najas flexilis	Slender naiad	46.6	△	-52.2	▽	-100.0	▽		△	152.4	△	-44.3	▽	137.0	△
	Fissidens spp. & Fontinalis spp.	Aquatic Moss	30.3	△	139.1	△	-29.0	▽	-100.0	▼		▲	-72.5	▼	196.2	▲
	Potamogeton vaseyi	Vasey's pondweed		-		▲	-100.0	▼		▲	-100.0	▼		▲	18.5	△
	Potamogeton diversifolius	Water-thread pondweed	-83.7	▼	-100.0	▽		-		-		△	-74.0	▽	137.0	△
	Isoetes spp.	Quillwort spp.		△	-100.0	▽		△	185.7	△	8.2	△	-53.2	▽	-100.0	▽
	Eleocharis robbinsii	Robbins' spikerush		-		-		-		▲	-100.0	▼		-		-
	Lobelia dortmanna	Water lobelia	-51.1	▽	91.3	△	-49.3	▽	471.4	△	-32.5	▽	-50.0	▽	-40.8	▽
	Sparganium angustifolium	Narrow-leaf bur-reed	-2.3	▽	-100.0	▽		-		-		-		-		-
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)		-		-		-		-		-		△	314.7	▲
	Potamogeton spirillus	Spiral-fruited pondweed		-		-		-		-		▲	-100.0	▼		△
	Elodea nuttallii	Slender waterweed		-		▲	-100.0	▼		-		-		-		-
	Eleocharis palustris	Creeping spikerush		-		-		-		-		△	-100.0	▽		-
	Juncus pelocarpus	Brown-fruited rush	-100.0	▽		-		-		△	-100.0	▽		△	-100.0	▽
	Sparganium eurycarpum	Common bur-reed		-		-		-		-		-		-		-
	Sagittaria rigida	Stiff arrowhead		-		-		-		-		-		-		-
	Stuckenia pectinata	Sago pondweed		-		-		-		△	-100.0	▽		-		-
	Potamogeton foliosus	Leafy pondweed		-		-		-		-		-		△	-100.0	▽
	Lemna trisulca	Forked duckweed		-		△	-100.0	▽		-		-		-		-
	Elatine minima	Waterwort		-		-		-		-		-		-		-
	Sparganium sp.	Bur-reed sp.		-		-		-		-		-		-		-
	Sagittaria graminea	Grass-leaved arrowhead		-		-		-		-		△	-100.0	▽		-
	Schoenoplectus subterminalis	Water bulrush		-		-		-		-		-		-		-
	Potamogeton gramineus	Variable-leaf pondweed		-		-		-		-		-		△	-100.0	▽
	Heteranthera dubia	Water stargrass		△	-100.0	▽		-		-		-		-		-
	Eriocaulon aquaticum	Pipewort		-		-		-		-		-		-		-

▲ or ▼ = Change Statistically Valid (Chi-square; α = 0.05)
△ or ▽ = Change Not Statistically Valid (Chi-square; α = 0.05)

E

APPENDIX E

Strategic Analysis of Aquatic Plant Management in Wisconsin (June 2019).

Extracted Supplemental Chapters:

3.3 (Herbicide Treatment)
3.4 (Physical Removal)
3.5 (Biological Control)

chemical curtains or adjuvants (weighting or sticking agents) may also help to maintain adequate CET, however more research is needed in this area.

This rapid dissipation of herbicide off of treatment areas is important for resource managers to consider in planning, as treating numerous targeted areas at a 'localized' scale may actually result in low-concentrations capable of having lakewide impacts as the herbicide dissipates off of the individual treatment sites. In general, if the percentage of treated areas to overall lake surface area is >5% and targeted areas are treated at relatively high 2,4-D concentrations (e.g., 2.0-4.0 ppm), then anticipated lakewide concentrations after dissipation should be calculated to determine the likelihood of lakewide effects (Nault et al. 2018).

Aquatic-use herbicides are commercially available in both liquid and granular forms. Successful target species control has been reported with both granular and liquid formulations. While there has been a commonly held belief that granular products are able to 'hold' the herbicide on site for longer periods of time, actual field comparisons between granular and liquid 2,4-D forms revealed that they dissipated similarly when applied at small-scale sites (Nault et al. 2015). In fact, liquid 2,4-D had higher initial observed water column concentrations than the granular form, but in the majority of cases concentrations of both forms decreased rapidly to below detection limits within several hours after treatment Nault et al. 2015). Likewise, according to United Phosphorus, Inc. (UPI), the sole manufacturer of endothall, the granular formulation of endothall does not hold the product in a specific area significantly longer than the liquid form (Jacob Meganck [UPI], *personal communication*).

In addition, the stratification of water and the formation of a thermal density gradient can confine the majority of applied herbicides in the upper, warmer water layer of deep lakes. In some instances, the entire lake water volume is used to calculate how much active ingredient should be applied to achieve a specific lakewide target concentration. However, if the volume of the entire lake is used to calculate application rates for stratified lakes, but the chemical only readily mixes into the upper water layer, the achieved lakewide concentration is likely to be much higher than the target concentration, potentially resulting in unanticipated adverse ecological impacts.

Because herbicides cannot be applied directly to specific submersed target plants, the dissipation of herbicide over the treatment area can lead to direct contact with non-target plants and animals. No herbicide is completely selective (i.e., effective specifically on only a single target species). Some plant species may be more susceptible to a given herbicide than others, highlighting the importance of choosing the appropriate herbicide, or other non-chemical management approach, to minimize potential non-target effects of treatment. There are many herbicides and plant species for which the CET relationship that would negatively affect the plant is unknown. This is particularly important in the case of rare, special concern, or threatened and endangered species. Additionally, loss of habitat following any herbicide treatment or other management technique may cause indirect reductions in populations of invertebrates or other organisms. Some organisms will only recolonize the managed areas as aquatic plants become re-established.

Below are reviews for the most commonly used herbicides for APM in Wisconsin. Much of the information here was pulled directly from DNR's APM factsheets (http://dnr.wi.gov/lakes/plants/factsheets/), which were compiled in 2012 using U.S. EPA herbicide product labels, U.S. Army Corps of Engineers reports, and communications with natural resource agencies in other northern, lake-rich states. These have been supplemented with more recent information from primary research publications.

Each pesticide has at least one mode of action which is the specific mechanism by which the active ingredient exerts a toxic effect. For example, some herbicides inhibit production of the pigments needed for photosynthesis while others mimic plant growth hormones and cause uncontrolled and unsustainable growth. Herbicides are often classified as either systemic or contact in mode of action, although some herbicides are able to function under various modes of action depending on environmental variables such as water temperature. Systemic pesticides are those that are absorbed by organisms and can be moved or translocated within the organism. Contact pesticides are those that exert toxic effects on the part(s) of an organism that they come in contact with. The amount of exposure time needed to kill an organism is based on the specific mode of action and the concentration of any given pesticide. In the descriptions below herbicides are generally categorized into which environment (above or below water) they are primarily used and a relative assessment of how quickly they impact plants. Herbicides can be applied in many ways. In lakes, they are usually applied to the water's surface (or below the water's surface) through controlled release by equipment including spreaders, sprayers, and underwater hoses. In wetland environments, spraying by helicopter, backpack sprayer, or application by cut-stem dabbing, wicking, injection, or basal bark application are also used.

S.3.3.1. Submersed or Floating, Relatively Fast-Acting Herbicides

Diquat

Registration and Formulations

Diquat (or diquat dibromide) initially received Federal registration for control of submersed and floating aquatic plants in 1962. It was initially registered with the U.S. EPA in 1986, evaluated for reregistration in 1995, and is currently under registration review. A registration review decision was expected in 2015 but has not been released (EPA Diquat Plan 2011). The active ingredient is 6,7-dihydrodipyrido[1,2-α:2',1'-c] pyrazinediium dibromide, and is commercially sold as liquid formulations for aquatic use.

Mode of Action and Degradation

Diquat is a fast-acting herbicide that works through contact with plant foliage by disrupting electron flow in photosystem I of the photosynthetic reaction, ultimately causing the destruction of cell membranes (Hess 2000; WSSA 2007). Plant tissues in contact with diquat become impacted within several hours after application, and within one to three days the plant tissue will become necrotic. Diquat is considered a non-selective herbicide and will rapidly kill a wide variety of plants on contact. Because diquat is a fast-acting herbicide, it is oftentimes used for managing plants growing in areas where water exchange is anticipated to limit herbicide exposure times, such as small-scale treatments.

Due to rapid vegetation decomposition after treatment, only partial treatments of a waterbody should be conducted to minimize dissolved oxygen depletion and associated negative impacts on fish and other aquatic organisms. Untreated areas can be treated with diquat 14 days after the first application.

Diquat is strongly attracted to silt and clay particles in the water and may not be very effective under highly turbid water conditions or where plants are covered with silt (Clayton and Matheson 2010).

The half-life of diquat in water generally ranges from a few hours to two days depending on water quality and other environmental conditions. Diquat has been detected in the water column from less than a day up towards 38 DAT, and remains in the water column longer when treating waterbodies with sandy sediments with lower organic matter and clay content (Coats et al. 1964; Grzenda et al. 1966; Yeo 1967; Sewell et al. 1970; Langeland and Warner 1986; Langeland et al. 1994; Poovey and Getsinger 2002; Parsons et al. 2007; Gorzerino et al. 2009; Robb et al. 2014). One study reported that diquat is chemically stable within a pH range of 3 to 8 (Florêncio et al. 2004). Due to the tendency of diquat to be rapidly adsorbed to suspended clays and particulates, long exposure periods are oftentimes not possible to achieve in the field. Studies conducted by Wersal et al. (2010a) did not observe differences in target species efficacy between daytime versus night-time applications of diquat. While large-scale diquat treatments are typically not implemented, a study by Parsons et al. (2007), observed declines in both dissolved oxygen and water clarity following the herbicide treatment.

Diquat binds indefinitely to organic matter, allowing it to accumulate and persist in the sediments over time (Frank and Comes 1967; Simsiman and Chesters 1976). It has been reported to have a very long-lived half-life (1000 days) in sediment because of extremely tight soil sorption, as well as an extremely low rate of degradation after association with sediment (Wauchope et al. 1992; Peterson et al. 1994). Both photolysis and microbial degradation are thought to play minor roles in degradation (Smith and Grove 1969; Emmett 2002). Diquat is not known to leach into groundwater due to its very high affinity to bind to soils.

One study reported that combinations of diquat and penoxsulam resulted in an antagonistic response between the herbicides when applied to water hyacinth (*Eichhornia crassipes*) and resulted in reduced efficacy than when applying penoxsulam alone. The antagonistic response is likely due to the rapid cell destruction by diquat that limits the translocation and efficacy of the slower acting enzyme inhibiting herbicides (Wersal and Madsen 2010b).

Toxicology

There are no restrictions on swimming or eating fish from waterbodies treated with diquat. Depending on the concentration applied, there is a 1-3 day waiting period after treatment for drinking water. However, in one study, diquat persisted in the water at levels above the EPA drinking water standard for at least 3 DAT, suggesting that the current 3-day drinking water restriction may not be sufficient under all application scenarios (Parsons et al. 2007). Water treated with diquat should not be used for pet or livestock drinking water for one day following treatment. The irrigation restriction for food crops is five days, and for ornamental plants or lawn/turf, it varies from one to three days depending on the concentration used. A study by Mudge et al. (2007)

on the effects of diquat on five popular ornamental plant species (begonia, dianthus, impatiens, petunia, and snapdragon) found minimal risks associated with irrigating these species with water treated with diquat up to the maximum use rate of 0.37 ppm.

Ethylene dibromide (EDB) is a trace contaminant in diquat products which originates from the manufacturing process. EDB is a documented carcinogen, and the EPA has evaluated the health risk of its presence in formulated diquat products. The maximum level of EDB in diquat dibromide is 0.01 ppm (10 ppb). EBD degrades over time, and it does not persist as an impurity.

Diquat does not have any apparent short-term effects on most aquatic organisms that have been tested at label application rates (EPA Diquat RED 1995). Diquat is not known to bioconcentrate in fish tissues. A study using field scenarios and well as computer modelling to examine the potential ecological risks posed by diquat determined that diquat poses a minimal ecological impact to benthic invertebrates and fish (Campbell et al. 2000). Laboratory studies indicate that walleye (*Sander vitreus*) are more sensitive to diquat than some other fish species, such as smallmouth bass (*Micropterus dolomieu),* largemouth bass (*Micropterus salmoides*), and bluegills (*Lepomis macrochirus*), with individuals becoming less sensitive with age (Gilderhus 1967; Paul et al. 1994; Shaw and Hamer 1995). Maximum application rates were lowered in response to these studies, such that applying diquat at recommended label rates is not expected to result in toxic effects on fish (EPA Diquat RED 1995). Sublethal effects such as respiratory stress or reduced swimming capacity have been observed in studies where certain fish species (e.g., yellow perch (*Perca flavescens*), rainbow trout (*Oncorhynchus mykiss*), and fathead minnows (*Pimephales promelas*)) have been exposed to diquat concentrations (Bimber et al. 1976; Dodson and Mayfield 1979; de Peyster and Long 1993). Another study showed no observable effects on eastern spiny softshell turtles (*Apalone spinifera spinifera*; Paul and Simonin 2007). Reduced size and pigmentation or increased mortality have been shown in some amphibians but at above recommended label rates (Anderson and Prahlad 1976; Bimber and Mitchell 1978; Dial and Bauer-Dial 1987). Toxicity data on invertebrates are scarce and diquat is considered not toxic to most of them. While diquat is not highly toxic to most invertebrates, significant mortality has been observed in some species at concentrations below the maximum label use rate for diquat, such as the amphipod *Hyalella azteca* (Wilson and Bond 1969; Williams et al. 1984), water fleas (*Daphnia* spp.). Reductions in habitat following treatment may also contribute to reductions of *Hyalella azteca*. For more information, a thorough risk assessment for diquat was compiled by the Washington State Department of Ecology Water Quality Program (WSDE 2002). Available toxicity data for fish, invertebrates, and aquatic plants is summarized in tabular format by Campbell et al. (2000).
Species Susceptibility

Diquat has been shown to control a variety of invasive submerged and floating aquatic plants, including Eurasian watermilfoil (*Myriophyllum spicatum*), curly-leaf pondweed (*Potamogeton crispus*), parrot feather (*Myriophyllum aquaticum*), Brazilian waterweed (*Egeria densa*), water hyacinth, water lettuce (*Pistia stratiotes*), flowering rush (*Butomus umbellatus*), and giant salvinia (*Salvinia molesta*; Netherland et al. 2000; Nelson et al. 2001; Poovey et al. 2002; Langeland et al. 2002; Skogerboe et al. 2006; Martins et al. 2007, 2008; Wersal et al. 2010a; Wersal and Madsen 2010a; Wersal and Madsen 2012; Poovey et al. 2012; Madsen et al. 2016). Studies conducted on the use of diquat for hydrilla (*Hydrilla verticillata*) and fanwort (*Cabomba caroliniana*) control have resulted in mixed reports of efficacy (Van et al. 1987; Langeland et al. 2002; Glomski et al. 2005; Skogerboe et al. 2006; Bultemeier et al. 2009; Turnage et al. 2015). Non-native phragmites (*Phragmites australis* subsp. *australis*) has been shown to not be significantly reduced by diquat (Cheshier et al. 2012).

Skogerboe et al. 2006 reported on the efficacy of diquat (0.185 and 0.37 ppm) under flow-through conditions (observed half-lives of 2.5 and 4.5 hours, respectively). All diquat treatments reduced Eurasian watermilfoil biomass by 97 to 100% compared to the untreated reference, indicating that this species is highly susceptible to diquat. Netherland et al. (2000) examined the role of various water temperatures (10, 12.5, 15, 20, and 25°C) on the efficacy of diquat applications for controlling curly-leaf pondweed. Diquat was applied at rates of 0.16-0.50 ppm, with exposure times of 9-12 hours. Diquat efficacy on curly-leaf pondweed was inhibited as water temperature decreased, although treatments at all temperatures were observed to significantly reduce biomass and turion formation. While the most efficacious curly-leaf pondweed treatments were conducted at 25°C, waiting until water warms to this temperature limits the potential for reducing turion production. Diquat applied at 0.37 ppm (with a 6 to 12-hour exposure time) or at 0.19 ppm (with a 72-hour exposure time) was effective at reducing biomass of flowering rush (Poovey et al. 2012; Madsen et al. 2016).

Native species that have been shown to be affected by diquat include: American lotus (*Nelumbo lutea*), common bladderwort (*Utricularia vulgaris*), coontail (*Ceratophyllum demersum*), common waterweed (*Elodea canadensis*), needle spikerush (*Eleocharis acicularis*), Illinois pondweed (*Potamogeton illinoensis*), leafy pondweed (*P. foliosus*), clasping-leaf pondweed (*P. richardsonii*), fern pondweed (*P. robbinsii*), sago pondweed (*Stuckenia pectinata*), and slender naiad (*Najas flexilis*) (Hofstra et al. 2001; Glomski et al. 2005; Skogerboe et al. 2006; Mudge 2013; Bugbee et al. 2015; Turnage et al. 2015). Diquat is particularly toxic to duckweeds (*Landoltia punctata* and *Lemna* spp.), although certain populations of dotted duckweed (*Landoltia punctata*) have developed resistance of diquat in waterbodies with a long history (20-30 years) of repeated diquat treatments (Peterson et al. 1997; Koschnick et al. 2006). Variable effects have been observed for water celery (*Vallisneria americana*), long-leaf pondweed (*Potamogeton nodosus*), and variable-leaf watermilfoil (*Myriophyllum heterophyllum*; Skogerboe et al. 2006; Glomski and Netherland 2007; Mudge 2013).

Flumioxazin

Registration and Formulations

Flumioxazin (2-[7-fluoro-3,4-dihydro-3-oxo-4-(2-propynyl)-2H-1,4-benzoxazin-6-yl]-4,5,6,7-tetrahydro-1H-isoindole-1,3(2H)-dione) was registered with the U.S. EPA for agricultural use in 2001 and registered for aquatic use in 2010. The first registration review of flumioxazin is expected to be completed in 2017 (EPA Flumioxazin Plan 2011). Granular and liquid formulations are available for aquatic use.

Mode of Action and Degradation

The mode of action of flumioxazin is through disruption of the cell membrane by inhibiting protoporphyrinogen oxidase which blocks production of heme and chlorophyll. The efficacy of this mode of action is dependent on both light intensity and water pH (Mudge et al. 2012a; Mudge and Haller 2010; Mudge et al. 2010), with herbicide degradation increasing with pH and efficacy decreasing as light intensity declines.

Flumioxazin is broken down by water (hydrolysis), light (photolysis) and microbes. The half-life ranges from approximately 4 days at pH 5 to 18 minutes at pH 9 (EPA Flumioxazin 2003). In the majority of Wisconsin lakes half-life should be less than 1 day.

Flumioxazin degrades into APF (6-amino-7-fluro-4-(2-propynyl)-1,4,-benzoxazin-3(2H)-one) and THPA (3,4,5,6-tetrahydrophthalic acid). Flumioxazin has a low potential to leach into groundwater due to the very quick hydrolysis and photolysis. APF and THPA have a high potential to leach through soil and could be persistent.

<u>Toxicology</u>

Tests on warm and cold-water fishes indicate that flumioxazin is "slightly to moderately toxic" to fish on an acute basis, with possible effects on larval growth below the maximum label rate of 0.4 ppm (400 ppb). Flumioxazin is moderately to highly toxic to aquatic invertebrates, with possible impacts below the maximum label rate. The potential for bioaccumulation is low since degradation in water is so rapid. The metabolites APF and THPA have not been assessed for toxicity or bioaccumulation.

The risk of acute exposure is primarily to chemical applicators. Concentrated flumioxazin doesn't pose an inhalation risk but can cause skin and eye irritation. Recreational water users would not be exposed to concentrated flumioxazin.

Acute exposure studies show that flumioxazin is "practically non-toxic" to birds and small mammals. Chronic exposure studies indicate that flumioxazin is non-carcinogenic. However, flumioxazin may be an endocrine disrupting compound in mammals (EPA Flumioxazin 2003), as some studies on small mammals did show effects on reproduction and larval development, including reduced offspring viability, cardiac and skeletal malformations, and anemia. It does not bioaccumulate in mammals, with the majority excreted in a week.

<u>Species Susceptibility</u>

The maximum target concentration of flumioxazin is 0.4 ppm (400 ppb). At least one study has shown that flumioxazin (at or below the maximum label rate) will control the invasive species fanwort (*Cabomba caroliniana*), hydrilla (*Hydrilla verticillata*), Japanese stiltgrass (*Microstegium vimineum*), Eurasian watermilfoil (*Myriophyllum spicatum*), water lettuce (*Pistia stratiotes*), curly-leaf pondweed (*Potamogeton crispus*), and giant salvinia (*Salvinia molesta*), while water hyacinth (*Eichhornia crassipes*) and water pennyworts (*Hydrocotyle* spp.) do not show significant impacts (Bultemeier et al. 2009; Glomski and Netherland 2013a; Glomski and Netherland 2013b; Mudge 2013; Mudge and Netherland 2014; Mudge and Haller 2012; Mudge and Haller 2010). Flowering rush (*Butomus umbellatus*; submersed form) showed mixed success in herbicide trials

(Poovey et al. 2012; Poovey et al. 2013). Native species that were significantly impacted (in at least one study) include coontail (*Ceratophyllum demersum*), water stargrass (*Heteranthera dubia*), variable-leaf watermilfoil (*Myriophyllum heterophyllum*), America lotus (*Nelumbo lutea*), pond-lilies (*Nuphar* spp.), white waterlily (*Nymphaea odorata*), white water crowfoot (*Ranunculus aquatilis*), and broadleaf cattail (*Typha latifolia*), while common waterweed (*Elodea canadensis*), squarestem spikerush (*Eleocharis quadrangulate*), horsetail (*Equisetum hyemale*), southern naiad (*Najas guadalupensis*), pickerelweed (*Pontederia cordata*), Illinois pondweed (*Potamogeton illinoensis*), long-leaf pondweed (*P. nodosus*), broadleaf arrowhead (*Sagittaria latifolia*), hardstem bulrush (*Schoenoplectus acutus*), common three-square bulrush (*S. pungens*), softstem bulrush (*S. tabernaemontani*), sago pondweed (*Stuckenia pectinata*), and water celery (*Vallisneria americana*) were not impacted relative to controls. Other species are likely to be susceptible, for which the effects of flumioxazin have not yet been evaluated.

Carfentrazone-ethyl

Registration and Formulations

Carfentrazone-ethyl is a contact herbicide that was registered with the EPA in 1998. The active ingredient is ethyl 2-chloro-3-[2 -chloro-4-fluoro-5-[4 -(difluoromethyl)-4,5-diydro-3-methyl-5-oxo-1H-1,2,4-trizol-1-yl)phenyl]propanoate. A liquid formulation of carfentrazone-ethyl is commercially sold for aquatic use.

Mode of Action and Degradation

Carfentrazone-ethyl controls plants through the process of membrane disruption which is initiated by the inhibition of the enzyme protoporphyrinogen oxidase, which interferes with the chlorophyll biosynthetic pathway. The herbicide is absorbed through the foliage of plants, with injury symptoms viable within a few hours after application, and necrosis and death observed in subsequent weeks.

Carfentrazone-ethyl breaks down rapidly in the environment, while its degradates are persistent in aquatic and terrestrial environments. The herbicide primarily degrades via chemical hydrolysis to carfentrazone-chloropropionic acid, which is then further degraded to carfentrazone -cinnamic, -propionic, -benzoic and 3-(hydroxymethyl)-carfentrazone-benzoic acids. Studies have shown that degradation of carfentrazone-ethyl applied to water (pH = 7-9) has a half-life range of 3.4-131 hours, with longer half-lives (>830 hours) documented in waters with lower pH (pH = 5). Extremes in environmental conditions such as temperature and pH may affect the activity of the herbicide, with herbicide symptoms being accelerated under warm conditions.

While low levels of chemical residue may occur in surface and groundwater, risk concerns to non-target organisms are not expected. If applied into water, carfentrazone-ethyl is expected to adsorb to suspended solids and sediment.

Toxicology

There is no restriction on the use of treated water for recreation (e.g., fishing and swimming). Carfentrazone-ethyl should not be applied directly to water within ¼ mile of an active potable water intake. If applied around or within potable water intakes, intakes must be turned off prior to application and remain turned off for a minimum of 24 hours following application; the intake may be turned on prior to 24 hours only if the carfentrazone-ethyl and major degradate level is determined by laboratory analysis to be below 200 ppb. Do not use water treated with carfentrazone-ethyl for irrigation in commercial nurseries or greenhouses. In scenarios where the herbicide is applied to 20% or more of the surface area, treated water should not be used for irrigation of crops until 14 days after treatment, or until the carfentrazone-ethyl and major degradate level is determined by analysis to be below 5 ppb.

In scenarios where the herbicide is applied as a spot treatment to less than 20% of the waterbody surface area, treated water may be used for irrigation by commercial turf farms and on residential turf and ornamentals without restriction. If more than 20% of the waterbody surface area is treated, water should not be used for irrigation of turf or ornamentals until 14 days after treatment, or until the carfentrazone-ethyl and major degradate level is determined by analysis to be below 5 ppb.

Carfentrazone-ethyl is listed as very toxic to certain species of algae and listed as moderately toxic to fish and aquatic animals. Treatment of dense plants beds may result in dissolved oxygen declines from plant decomposition which may lead to fish suffocation or death. To minimize impacts, applications of this herbicide should treat up to a maximum of half of the waterbody at a time and wait a minimum of 14 days before retreatment or treatment of the remaining half of the waterbody. Carfentrazone-ethyl is considered to be practically non-toxic to birds on an acute and sub-acute basis.

Carfentrazone-ethyl is harmful if swallowed and can be absorbed through the skin or inhaled. Those who mix or apply the herbicide need to protect their skin and eyes from contact with the herbicide to minimize irritation and avoid breathing the spray mist. Carfentrazone-ethyl is not carcinogenic, neurotoxic, or mutagenic and is not a developmental or reproductive toxicant.

Species Susceptibility

Carfentrazone-ethyl is used for the control of floating and emergent aquatic plants such as duckweeds (*Lemna* spp.), watermeals (*Wolffia* spp.), water lettuce (*Pistia stratiotes*), water hyacinth (*Eichhornia crassipes*), and salvinia (*Salvinia* spp.). Carfentrazone-ethyl can also be used to control submersed plants such as Eurasian watermilfoil (*Myriophyllum spicatum*).

S.3.3.2. Submersed, Relatively Slow-Acting Herbicides

2,4-D

Registration and Formulations

2,4-D is an herbicide that is widely used as a household weed-killer, agricultural herbicide, and aquatic herbicide. It has been in use since 1946 and was registered with the U.S. EPA in 1986 and evaluated and reregistered in 2005. It is currently being evaluated for reregistration, and the estimated registration review decision date was in 2017 (EPA 2,4-D Plan 2013). The active ingredient is 2,4-dichloro-phenoxyacetic acid. There are two types of 2,4-D used as aquatic herbicides: dimethyl amine salt (DMA) and butoxyethyl ester (BEE). The ester formulations are toxic to fish and some important invertebrates such as water fleas (*Daphnia* spp.) and midges at application rates. 2,4-D is commercially sold as a liquid amine as well as ester and amine granular products for control of submerged, emergent, and floating-leaf vegetation. Only 2,4-D products labeled for use in aquatic environments may be used to control aquatic plants.

Mode of Action and Degradation

Although the exact mode of action of 2,4-D is not fully understood, the herbicide is traditionally believed to target broad-leaf dicotyledon species with minimal effects generally observed on numerous monocotyledon species, especially in terrestrial applications (WSSA 2007). 2,4-D is a systemic herbicide which affects plant cell growth and division. Upon application, it mimics the natural plant hormone auxin, resulting in bending and twisting of stems and petioles followed by growth inhibition, chlorosis (reduced coloration) at growing points, and necrosis or death of sensitive species (WSSA 2007). Following treatment, 2,4-D is taken up by the plant and translocated through the roots, stems and leaves, and plants begin to die within one to two weeks after application, but can take several weeks to decompose. The total length of target plant roots can be an important in determining the response of an aquatic plant to 2,4-D (Belgers et al. 2007). Treatments should be made when plants are growing. After treatment, the 2,4-D concentration in the water is reduced primarily through microbial activity, off-site movement by water, or adsorption to small particles in silty water.

Previous studies have indicated that 2,4-D degradation in water is highly variable depending on numerous factors such as microbial presence, temperature, nutrients, light, oxygen, organic content of substrate, pH, and whether or not the water has been previously exposed to 2,4-D or other phenoxyacetic acids (Howard et al. 1991). Once in contact with water, both the ester and amine formulations dissociate to the acid form of 2,4-D, with a faster dissociation to the acid form under more alkaline conditions. 2,4-D degradation products include 1,2,4-benzenetriol, 2,4-dichlorophenol, 2,4-dichloroanisole, chlorohydroquinone (CHQ), 4-chlorophenol, and volatile organics.

The half-life of 2,4-D has a wide range depending on water conditions. Half-lives have been reported to range from 12.9 to 40 days, while in anaerobic lab conditions the half-life has been measured at 333 days (EPA RED 2,4-D 2005). In large-scale low-concentration 2,4-D treatments monitored across numerous Wisconsin lakes, estimated half-lives ranged from 4-76 days, and the rate of herbicide degradation was generally observed to be slower in oligotrophic seepage lakes. Of these large-scale 2,4-D treatments, the threshold for irrigation of plants which are not labeled for direct treatment with 2,4-D (<0.1 ppm (100 ppb) by 21 DAT) was exceeded the majority of the treatments (Nault et al. 2018). Previous historical use of 2,4-D may also be an important variable to consider, as microbial communities which are responsible for the breakdown of 2,4-D may potentially exhibit changes in community composition over time with repeated use (de Lipthay et al. 2003; Macur et al. 2007). Additional detailed information on the environmental fate of 2,4-D is compiled by Walters 1999.

There have been some preliminary investigations into the concentration of primarily granular 2,4-D in water-saturated sediments, or pore-water. Initial results suggest the concentration of 2,4-D in the pore-water varies widely from site to site following a chemical treatment, although in some locations the concentration in the pore-water was observed to be 2-3 times greater than the application rate (Jim Kreitlow [DNR], *personal communication*). Further research and additional studies are needed to assess the implications of this finding for target species control and non-target impacts on a variety of organisms.

<u>Toxicology</u>

There are no restrictions on eating fish from treated waterbodies, human drinking water, or pet/livestock drinking water. Based upon 2,4-D ester (BEE) product labels, there is a 24-hour waiting period after treatment for swimming. Before treated water can be used for irrigation, the concentration must be below 0.1 ppm (100 ppb), or at least 21 days must pass. Adverse health effects can be produced by acute and chronic exposure to 2,4-D. Those who mix or apply 2,4-D need to protect their skin and eyes from contact with 2,4-D products to minimize irritation and avoid inhaling the spray. In its consideration of exposure risks, the EPA believes no significant risks will occur to recreational users of water treated with 2,4-D.

There are differences in toxicity of 2,4-D depending on whether the formulation is an amine (DMA) or ester (BEE), with the BEE formulation shown to be more toxic in aquatic environments. BEE formulations are considered toxic to fish and invertebrates such as water fleas and midges at operational application rates. DMA formulations are not considered toxic to fish or invertebrates at operational application rates. Available data indicate 2,4-D does not accumulate at significant levels in the tissues of fish. Although fish exposed to 2,4-D may take up very small amounts of its breakdown products to then be metabolized, the vast majority of these products are rapidly excreted in urine (Ghassemi et al. 1981).

On an acute basis, EPA assessment considers 2,4-D to be “practically non-toxic” to honeybees and tadpoles. Dietary tests (substance administered in the diet for five consecutive days) have shown 2,4-D to be “practically non-toxic” to birds, with some species being more sensitive than others (when 2,4-D was orally and directly administered to birds by capsule or gavage, the substance was “moderately toxic” to some species). For freshwater invertebrates, EPA considers 2,4-D amine to be “practically non-toxic” to “slightly toxic” (EPA RED 2,4-D 2005). Field studies on the potential impact of 2,4-D on benthic macroinvertebrate communities have generally not observed significant changes, although at least one study conducted in Wisconsin observed negative correlations in macroinvertebrate richness and abundance following treatment, and further studies are likely warranted (Stephenson and Mackie 1986; Siemering et al. 2008; Harrahy et al. 2014). Additionally, sublethal effects such as mouthpart deformities and change in sex ratio have been observed in the midge *Chironomus riparius* (Park et al. 2010).

While there is some published literature available looking at short-term acute exposure of various aquatic organisms to 2,4-D, there is limited literature is available on the effects of low-concentration chronic exposure to commercially available 2,4-D formulations (EPA RED 2,4-D 2005). The department recently funded several projects related to increasing our understanding of the potential impacts of chronic exposure to low-concentrations of 2,4-D through AIS research and development grants. One of these studies observed that fathead minnows (*Pimephales promelas*) exposed under laboratory conditions for 28 days to 0.05 ppm (50 ppb) of two different commercial formulations of 2,4-D (DMA® 4 IVM and Weedestroy® AM40) had decreases in larval survival and tubercle presence in males, suggesting that these formulations may exert some degree of chronic toxicity or endocrine-disruption which has not been previously observed when testing pure compound 2,4-D (DeQuattro and Karasov 2016). However, another follow-up study determined that fathead minnow larval survival (30 days post hatch) was decreased following exposure of eggs and larvae to pure 2,4-D, as well as to the two commercial formulations (DMA® 4 IVM and Weedestroy® AM40), and also identified a critical window of exposure for effects on survival to the period between fertilization and 14 days post hatch (Dehnert et al. 2018).

Another related follow-up laboratory study is currently being conducted to examine the effects of 2,4-D exposure on embryos and larvae of several Wisconsin native fish species. Preliminary results indicate that negative impacts of embryo survival were observed for 4 of the 9 native species tested (e.g., walleye, northern pike, white crappie, and largemouth bass), and negative impacts of larval survival were observed for 4 of 7 natives species tested (e.g., walleye, yellow perch, fathead minnows, and white suckers; Dehnert and Karasov, *in progress*).

A controlled field study was conducted on six northern Wisconsin lakes to understand the potential impacts of early season large-scale, low-dose 2,4-D on fish and zooplankton (Rydell et al. 2018). Three lakes were treated with early season low-dose liquid 2,4-D (lakewide epilimnetic target rate: 0.3 ppm (300 ppb)), while the other three lakes served as reference without treatment. Zooplankton densities were similar within lakes during the pre-treatment year and year of treatment, but different trends in several zooplankton species were observed in treatment lakes during the year following treatment. Peak abundance of larval yellow perch (*Perca flavescens*) was lower in the year following treatment, and while this finding was not statistically significant, decreased larval yellow perch abundance was not observed in reference lakes. The observed declines in larval yellow perch abundance and changes in zooplankton trends within treatment lakes in the year after treatment may be a result of changes in aquatic plant communities and not a direct effect of treatment. No significant effect was observed on peak abundance of larval largemouth bass (*Micropterus salmoides*), minnows, black crappie (*Pomoxis nigromaculatus*), bluegill (*Lepomis macrochirus*), or juvenile yellow perch. Larval black crappie showed no detectable response in growth or feeding success. Net pen trials for juvenile bluegill indicated no significant difference in survival between treatment and reference trials, indicating that no direct mortality was associated with the herbicide treatments. Detection of the level of larval fish mortality found in the lab studies would not have been possible in the field study given large variability in larval fish abundance among lakes and over time.

Concerns have been raised about exposure to 2,4-D and elevated cancer risk. Some epidemiological studies have found associations between 2,4-D and increased risk of non-Hodgkin lymphoma in high exposure populations, while other studies have shown that increased cancer risk may be caused by other factors (Hoar et al. 1986; Hardell and Eriksson 1999; Goodman et al. 2015). The EPA determined in 2005 that there is not sufficient evidence to classify 2,4-D as a human carcinogen (EPA RED 2,4-D 2005).

Another chronic health concern with 2,4-D is the potential for endocrine disruption. There is some evidence that 2,4-D may have effects on reproductive development, though other studies suggest the findings may have had other causes (Garry et al. 1996; Coady et al. 2013; Goldner et al. 2013; Neal et al. 2017). The extent and implications of this are not clear and it is an area of ongoing research.

Detailed literature reviews of 2,4-D toxicology have been compiled by Garabrant and Philbert (2002), Jervais et al. (2008), and Burns and Swaen (2012).

<u>Species Susceptibility</u>

With appropriate concentration and exposure, 2,4-D is capable of reducing abundance of the invasive plant species Eurasian watermilfoil (*Myriophyllum spicatum*), parrot feather (*M. aquaticum*), water chestnut (*Trapa natans*), water hyacinth (*Eichhornia crassipes*), and water lettuce (*Pistia stratiotes*; Elliston and Steward 1972; Westerdahl et al. 1983; Green and Westerdahl 1990; Helsel et al. 1996, Poovey and Getsinger 2007; Wersal et al. 2010b; Cason and Roost 2011; Robles et al. 2011; Mudge and Netherland 2014). Perennial pepperweed (*Lepidium latifolium)* and fanwort (*Cabomba caroliniana*) have been shown to be somewhat tolerant of 2,4-D (Bultemeier et al. 2009; Whitcraft and Grewell 2012).

Efficacy and selectivity of 2,4-D is a function of concentration and exposure time (CET) relationships, and rates of 0.5-2.0 ppm coupled with exposure times ranging from 12 to 72 hours have been effective at achieving Eurasian watermilfoil control under laboratory settings (Green and Westerdahl 1990). In addition, long exposure times (>14 days) to low-concentrations of 2,4-D (0.1-0.25 ppm) have also been documented to achieve milfoil control (Hall et al. 1982; Glomski and Netherland 2010).

According to product labels, desirable native species that may be affected include native milfoils (*Myriophyllum* spp.), coontail (*Ceratophyllum demersum*), common waterweed (*Elodea canadensis*), naiads (*Najas* spp.), waterlilies (*Nymphaea* spp. and *Nuphar* spp.), bladderworts (*Utricularia* spp.), and duckweeds (*Lemna* spp.). While it may affect softstem bulrush (*Schoenoplectus tabernaemontani*), other species such as American bulrush (*Schoenoplectus americanus*) and muskgrasses (*Chara* spp.) have been shown to be somewhat tolerant of 2,4-D (Miller and Trout 1985; Glomski et al. 2009; Nault et al. 2014; Nault et al. 2018).

In large-scale, low-dose (0.073-0.5 ppm) 2,4-D treatments evaluated by Nault et al. (2018), milfoil exhibited statistically significant lakewide decreases in posttreatment frequency across 23 of the 28 (82%) of the treatments monitored. In lakes where year of treatment milfoil control was achieved, the longevity of control ranged from 2–8 years. However, it is important to note that milfoil was not 'eradicated' from any of these lakes and is still present even in those lakes which have sustained very low frequencies over time. While good year of treatment control was achieved in all lakes with pure Eurasian watermilfoil populations, significantly reduced control was observed in the majority of lakes with hybrid watermilfoil (*Myriophyllum spicatum* x *sibiricum*) populations. Eurasian watermilfoil control was correlated with the mean concentration of 2,4-D measured during the first two weeks of treatment, with increasing lakewide concentrations resulting in increased Eurasian watermilfoil control. In contrast, there was no significant relationship observed between Eurasian watermilfoil control and mean concentration of 2,4-D. In lakes where good (>60%) year of treatment control of hybrid watermilfoil was achieved, 2,4-D degradation was slow, and measured lakewide concentrations were sustained at >0.1 ppm (>100 ppb) for longer than 31 days. In addition to reduced year of treatment efficacy, the longevity of control was generally shorter in lakes that contained hybrid watermilfoil versus Eurasian watermilfoil, suggesting that hybrid watermilfoil may have the ability to rebound quicker after large-scale treatments than pure Eurasian watermilfoil populations. However, it is important to keep in mind that hybrid watermilfoil is broad term for multiple different strains, and variation in herbicide response and growth between specific genotypes of hybrid watermilfoil has been documented (Taylor et al. 2017).

In addition, the study by Nault et al. (2018) documented several native monocotyledon and dicotyledon species that exhibited significant declines posttreatment. Specifically, northern watermilfoil (*Myriophyllum sibiricum*), slender naiad (*Najas flexilis*), water marigold (*Bidens beckii*), and several thin-leaved pondweeds (*Potamogeton pusillus*, *P. strictifolius*, *P. friesii* and *P. foliosus*) showed highly significant declines in the majority of the lakes monitored. In addition, variable/Illinois pondweed (*P. gramineus*/*P. illinoensis*), flat-stem pondweed (*P. zosteriformis*), fern pondweed (*P. robbinsii*), and sago pondweed (*Stuckenia pectinata*) also declined in many lakes. Ribbon-leaf pondweed (*P. epihydrus*) and water stargrass (*Heteranthera dubia*) declined in the lakes where they were found. Mixed effects of treatment were observed with water celery (*Vallisneria americana*) and southern naiad (*Najas guadalupensis*), with some lakes showing significant declines posttreatment and other lakes showing increases.

Since milfoil hybridity is a relatively new documented phenomenon (Moody and Les 2002), many of the early lab studies examining CET for milfoil control did not determine if they were examining pure Eurasian watermilfoil or hybrid watermilfoil (*M. spicatum* x *sibiricum*) strains. More recent laboratory and mesocosm studies have shown that certain strains of hybrid watermilfoil exhibit more aggressive growth and are less affected by 2,4-D (Glomski and Netherland 2010; LaRue et al. 2013; Netherland and Willey 2017; Taylor et al. 2017), while other studies have not seen differences in overall growth patterns or treatment efficacy when compared to pure Eurasian watermilfoil (Poovey et al. 2007). Differences between Eurasian and hybrid watermilfoil control following 2,4-D applications have also been documented in the field, with lower efficacy and shorter longevity of hybrid watermilfoil control when compared to pure Eurasian watermilfoil populations (Nault et al. 2018). Field studies conducted in the Menominee River Drainage in northeastern Wisconsin and upper peninsula of Michigan observed hybrid milfoil genotypes more frequently in lakes that had previous 2,4-D treatments, suggesting possible selection of more tolerant hybrid strains over time (LaRue 2012).

Fluridone

Registration and Formulations

Fluridone is an aquatic herbicide that was initially registered with the U.S. EPA in 1986. It is currently being evaluated for reregistration. The estimated registration review decision date was in 2014 (EPA Fluridone Plan 2010). The active ingredient is (1-methyl-3-phenyl-5-[3-(trifluoromethyl) phenyl]-4(1H)-pyridinone). Fluridone is available in both liquid and slow-release granular formulations.

Mode of Action and Degradation

Fluridone's mode of action is to reduce a plant's ability to protect itself from sun damage. The herbicide prevents the plant from making a protective pigment and as a result, sunlight causes the plant's chlorophyll to break down. Treated plants will turn white or pink at the growing tips a week after exposure and will begin to die one to two months after treatment (Madsen et al. 2002). Therefore, fluridone is only effective if plants are actively growing at the time of treatment. Effective use of fluridone requires low, sustained concentrations and a relatively long contact time (e.g., 45-90 days). Due to this requirement, fluridone is usually applied to an entire waterbody or basin. Some success has been demonstrated when additional follow-up 'bump' treatments are used to maintain the low concentrations over a long enough period of time to produce control. Fluridone has also been applied to riverine systems using a drip system to maintain adequate CET.

Following treatment, the amount of fluridone in the water is reduced through dilution and water movement, uptake by plants, adsorption to the sediments, and via breakdown caused by light and microbes. Fluridone is primarily degraded through photolysis (Saunders and Mosier 1983), while depth, water clarity and light penetration can influence degradation rates (Mossler et al. 1989; West et al. 1983). There are two major degradation products from fluridone: n-methyl formamide (NMF) and 3-trifluoromethyl benzoic acid.

The half-life of fluridone can be as short as several hours, or hundreds of days, depending on conditions (West et al. 1979; West et al. 1983; Langeland and Warner 1986; Fox et al. 1991, 1996; Jacob et al. 2016). Preliminary work on a seepage lake in Waushara County, WI detected fluridone in the water nearly 400 days following an initial application that was then augmented to maintain concentrations via a 'bump' treatment at 60 and 100 days later (Onterra 2017a). Light exposure is influential in controlling degradation rate, with a half-life ranging from 15 to 36 hours when exposed to the full spectrum of natural sunlight (Mossler et al. 1989). As light wavelength increases, the half-life increases too, indicating that season and timing may affect fluridone persistence. Fluridone half-life has been shown to be only slightly dependent on fluridone concentration, oxygen concentration, and pH (Saunders and Mosier 1983). One study found that the half-life of fluridone in water was slightly lower when the herbicide was applied to the surface of the water as opposed to a sub-surface application, suggesting that degradation may also be affected by mode of application (West and Parka 1981).

The persistence of herbicide in the sediment has been reported to be much longer than in the overlying water column, with studies showing persistence ranges from 3 months to a year in

sediments (Muir et al. 1980; Muir and Grift 1982; West et al. 1983). Persistence in soil is influenced by soil chemistry (Shea and Weber 1983; Mossler et al. 1993). Fluridone concentrations measured in sediments reach a maximum in one to four weeks after treatment and decline in four months to a year depending on environmental conditions. Fluridone adsorbs to clay and soils with high organic matter, especially in pellet form, and can reduce the concentration of fluridone in the water. Adsorption to the sediments is reversible; fluridone gradually dissipates back into the water where it is subject to chemical breakdown.
Some studies have shown variable release time of the herbicide among different granular fluridone products (Mossler et al. 1993; Koschnick et al. 2003; Bultemeier and Haller 2015). In addition, pelletized formulations may be more effective in sandy hydrosoils, while aqueous suspension formulations may be more appropriate for areas with high amounts of clay or organic matter (Mossler et al. 1993)

Toxicology

Fluridone does not appear to have short-term or long-term effects on fish at approved application rates, but fish exposed to water treated with fluridone do absorb fluridone into their tissues. However, fluridone has demonstrated a very low potential for bioconcentration in fish, zooplankton, and aquatic plants (McCowen et al. 1979; West et al. 1979; Muir et al. 1980; Paul et al. 1994). Fluridone concentrations in fish decrease as the herbicide disappears from the water. Studies on the effects of fluridone on aquatic invertebrates (e.g., midge and water flea) have shown increased mortality at label application rates (Hamelink et al. 1986; Yi et al. 2011). Studies on birds indicate that fluridone would not pose an acute or chronic risk to birds. In addition, no treatment related effects were noted in mice, rats, and dogs exposed to dietary doses. No studies have been published on amphibians or reptiles. There are no restrictions on swimming, eating fish from treated waterbodies, human drinking water or pet/livestock drinking water. Depending on the type of waterbody treated and the type of plant being watered, irrigation restrictions may apply for up to 30 days. There is some evidence that the fluridone degradation product NMF causes birth defects, though NMF has only been detected in the lab and not following actual fluridone treatments in the field, including those at maximum label rate (Osborne et al. 1989; West et al. 1990).

Species Susceptibility

Because fluridone treatments are often applied at a lakewide scale and many plant species are susceptible to fluridone, careful consideration should be given to potential non-target impacts and changes in water quality in response to treatment. Sustained native plant species declines and reductions in water clarity have been observed following fluridone treatments in field applications (O'Dell et al. 1995; Valley et al. 2006; Wagner et al. 2007; Parsons et al. 2009). However, reductions in water clarity are not always observed and can be avoided (Crowell et al. 2006). Additionally, the selective activity of fluridone is primarily rate-dependent based on analysis of pigments in nine aquatic plant species (Sprecher et al. 1998b).

Fluridone is most often used for control of invasive species such as Eurasian and hybrid watermilfoil (*Myriophyllum spicatum* x *sibiricum*), Brazilian waterweed (*Egeria densa*), and hydrilla (*Hydrilla verticillata*; Schmitz et al. 1987; MacDonald et al. 1993; Netherland et al. 1993;

Netherland and Getsinger 1995a, 1995b; Cockreham and Netherland 2000; Hofstra and Clayton 2001; Madsen et al. 2002; Netherland 2015). However, fluridone tolerance has been observed in some hydrilla and hybrid watermilfoil populations (Michel et al. 2004; Arias et al. 2005; Puri et al. 2006; Slade et al. 2007; Berger et al. 2012, 2015; Thum et al. 2012; Benoit and Les 2013; Netherland and Jones 2015). Fluridone has also been shown to affect flowering rush (*Butomus umbellatus*), fanwort (*Cabomba caroliniana*), buttercups (*Ranunculus* spp.), long-leaf pondweed (*Potamogeton nodosus*), Illinois pondweed (*P. illinoensis*), leafy pondweed (*P. foliosus*), flat-stem pondweed (*P. zosteriformis*), sago pondweed (*Stuckenia pectinata*), oxygen-weed (*Lagarosiphon major*), northern watermilfoil (*Myriophyllum sibiricum*), variable-leaf watermilfoil (*M. heterophyllum*), curly-leaf pondweed (*Potamogeton crispus*), coontail (*Ceratophyllum demersum*), common waterweed (*Elodea canadensis*), southern naiad (*Najas guadalupensis*), slender naiad (*N. flexilis*), white waterlily (*Nymphaea odorata*), water marigold (*Bidens beckii*), duckweed (*Lemna* spp.), and watermeal (*Wolffia columbiana*) (Wells et al. 1986; Kay 1991; Farone and McNabb 1993; Netherland et al. 1997; Koschnick et al. 2003; Crowell et al. 2006; Wagner et al. 2007; Parsons et al. 2009; Cheshier et al. 2011; Madsen et al. 2016). Muskgrasses (*Chara* spp.), water celery (*Vallisneria americana*), cattails (*Typha* spp.), and willows (*Salix* spp.) have been shown to be somewhat tolerant of fluridone (Farone and McNabb 1993; Poovey et al. 2004; Crowell et al. 2006).

Large-scale fluridone treatments that targeted Eurasian and hybrid watermilfoils have been conducted in several Wisconsin lakes. Recently, five of these waterbodies treated with low-dose fluridone (2-4 ppb) have been tracked over time to understand herbicide dissipation and degradation patterns, as well as the efficacy, selectivity, and longevity of these treatments. These field trials resulted in a pre- vs. post-treatment decrease in the number of vegetated littoral zone sampling sites, with a 9-26% decrease observed following treatment (an average decrease in vegetated littoral zone sites of 17.4% across waterbodies). In four of the five waterbodies, substantial decreases in plant biomass (≥10% reductions in average total rake fullness) was documented at sites where plants occurred in both the year of and year after treatment. Good milfoil control was achieved, and long-term monitoring is ongoing to understand the longevity of target species control over time. However, non-target native plant populations were also observed to be negatively impacted in conjunction with these treatments, and long-term monitoring is ongoing to understand their recovery over time. Exposure times in the five waterbodies monitored were found to range from 320 to 539 days before falling below detectable limits. Data from these recent projects is currently being compiled and a compressive analysis and report is anticipated in the near future.

Endothall

Registration and Formulations

Endothall was registered with the U.S. EPA for aquatic use in 1960 and reregistered in 2005 (Menninger 2012). Endothall is the common name of the active ingredient endothal acid (7-oxabicyclo[2,2,1] heptane-2,3-dicarboxylic acid). Granular and liquid formulations are currently registered by EPA and DATCP. Endothall products are used to control a wide range of terrestrial and aquatic plants. Two types of endothall are available: dipotassium salt and dimethylalkylamine salt (“mono-N,N-dimethylalkylamine salt” or “monoamine salt”). The dimethylalkylamine salt form is toxic to fish and other aquatic organisms and is faster-acting than the dipotassium salt form.

Mode of Action and Degradation

Endothall is considered a contact herbicide that inhibits respiration, prevents the production of proteins and lipids, and disrupts the cellular membrane in plants (MacDonald et al. 1993; MacDonald et al. 2001; EPA RED Endothall 2005; Bajsa et al. 2012). Although typical rates of endothall application inhibit plant respiration, higher concentrations have been shown to increase respiration (MacDonald et al. 2001). The mode of action of endothall is unlike any other commercial herbicide. For effective control, endothall should be applied when plants are actively growing, and plants begin to weaken and die within a few days after application.

Uptake of endothall is increased at higher water temperatures and higher amounts of light (Haller and Sutton 1973). Netherland et al. (2000) found that while biomass reduction of curly-leaf pondweed (*Potamogeton crispus*) was greater at higher water temperature, reductions of turion production were much greater when curly-leaf pondweed was treated a lower water temperature (18 °C vs 25 °C).

Degradation of endothall is primarily microbial (Sikka and Saxena 1973) and half-life of the dipotassium salt formulations is between 4 to 10 days (Reinert and Rodgers 1987; Reynolds 1992), although dissipation due to water movement may significantly shorten the effective half-life in some treatment scenarios. Half of the active ingredient from granular endothall formulations has been shown to be released within 1-5 hours under conditions that included water movement (Reinert et al. 1985; Bultemeier and Haller 2015). Endothall is highly water soluble and does not readily adsorb to sediments or lipids (Sprecher et al. 2002; Reinert and Rodgers 1984). Degradation from sunlight or hydrolysis is very low (Sprecher et al. 2002). The degradation rate of endothall has been shown to increase with increasing water temperature (UPI, *unpublished data*). The degradation rate is also highly variable across aquatic systems and is much slower under anaerobic conditions (Simsiman and Chesters 1975). Relative to other herbicides, endothall is unique in that is comprised of carbon, hydrogen, and oxygen with the addition of potassium and nitrogen in the dipotassium and dimethylalkylamine formulations, respectively. This allows for complete breakdown of the herbicide without additional intermediate breakdown products (Sprecher et al. 2002).

Toxicology

All endothall products have a drinking water standard of 0.1 ppm and cannot be applied within 600 feet of a potable water intake. Use restrictions for dimethylalkylamine salt formulations have additional irrigation and aquatic life restrictions.

Dipotassium salt formulations

At recommended rates, the dipotassium salt formulations appear to have few short-term behavioral or reproductive effects on bluegill (*Lepomis macrochirus*) or largemouth bass (*Micropterus salmoides;* Serns 1977; Bettolli and Clark 1992; Maceina et al. 2008). Bioaccumulation of dipotassium salt formulations by fish from water treated with the herbicide is unlikely, with studies showing less than 1% of endothall being taken up by bluegill (Sikka et al. 1975; Serns 1977). In addition, studies have shown the dipotassium salt formulation induces no significant adverse effects on aquatic invertebrates when used at label application rates (Serns 1975; Williams et al. 1984). A freshwater mussel species was found to be more sensitive to dipotassium salt endothall than other invertebrate species tested, but significant acute toxicity was still only found at concentrations well above the maximum label rate. However, as with other plant control approaches, some aquatic plant-dwelling populations of aquatic organisms may be adversely affected by application of endothall formulations due to habitat loss.

During EPA reregistration of endothall in 2005, it was required that product labels state that lower rates of endothall should be used when treating large areas, "such as coves where reduced water movement will not result in rapid dilution of the herbicide from the target treatment area or when treating entire lakes or ponds."

Dimethylalkylamine salt formulations

In contrast to the respective low to slight toxicity of the dipotassium salt formulations to fish and aquatic invertebrates, laboratory studies have shown the dimethylalkylamine formulations are toxic to fish and macroinvertebrates at concentrations above 0.3 ppm. In particular, the liquid formulation will readily kill fish present in a treatment site. Product labels for the dimethylalkylamine salt formulations recommend no treatment where fish are an important resource.

The dimethylalkylamine formulations are more active on aquatic plants than the dipotassium formulations, but also are 2-3 orders of magnitude more toxic to non-target aquatic organisms (EPA RED Endothall 2005; Keckemet 1969). The 2005 reregistration decision document limits aquatic use of the dimethylalkylamine formulations to algae, Indian swampweed (*Hygrophila polysperma*), water celery (*Vallisneria americana*), hydrilla (*Hydrilla verticillata*), fanwort (*Cabomba caroliniana*), bur reed (*Sparganium* sp.), common waterweed (*Elodea canadensis*), and Brazilian waterweed (*Egeria densa*). Coontail (*Ceratophyllum demersum*), watermilfoils (*Myriophyllum* spp.), naiads (*Najas* spp.), pondweeds (*Potamogeton* spp.), water stargrass (*Heteranthera dubia*), and horned pondweed (*Zannichellia palustris*) were to be removed from product labels (EPA RED Endothall 2005).

<u>Species Susceptibility</u>

According to the herbicide label, the maximum target concentration of endothall is 5000 ppb (5.0 ppm) acid equivalent (ae). Endothall is used to control a wide range of submersed species, including non-native species such as curly-leaf pondweed and Eurasian watermilfoil (*Myriophyllum spicatum*). The effects of the different formulations of endothall on various species of aquatic plants are discussed below.

Dipotassium salt formulations

At least one mesocosm or lab study has shown that endothall (at or below the maximum label rate) will control the invasive species hydrilla (Netherland et al. 1991; Wells and Clayton 1993; Hofstra and Clayton 2001; Pennington et al. 2001; Skogerboe and Getsinger 2001; Shearer and Nelson 2002; Netherland and Haller 2006; Poovey and Getsinger 2010), oxygen-weed (*Lagarosiphon major*; Wells and Clayton 1993; Hofstra and Clayton 2001), Eurasian watermilfoil (Netherland et al. 1991; Skogerboe and Getsinger 2002; Mudge and Theel 2011), water lettuce (*Pistia stratiotes*; Conant et al. 1998), curly-leaf pondweed (Yeo 1970), and giant salvinia (*Salvinia molesta*; Nelson et al. 2001). Wersal and Madsen (2010a) found that parrot feather (*Myriophyllum aquaticum*) control with endothall was less than 40% even with two days of exposure time at the maximum label rate. Endothall was shown to control the shoots of flowering rush (*Butomus umbellatus*), but control of the roots was variable (Poovey et al. 2012; Poovey et al. 2013). One study found that endothall did not significantly affect photosynthesis in fanwort with 6 days of exposure at 2.12 ppm ae (2120 ppb ae; Bultemeier et al. 2009). Large-scale, low-dose endothall treatments were found to reduce curly-leaf pondweed frequency, biomass, and turion production substantially in Minnesota lakes, particularly in the first 2-3 years of treatments (Johnson et al. 2012).

Native species that were significantly impacted (at or below the maximum endothall label rate in at least one mesocosm or lab study) include coontail (Yeo 1970; Hofstra and Clayton 2001; Hofstra et al. 2001; Skogerboe and Getsinger 2002; Wells and Clayton 1993; Mudge 2013), southern naiad (*Najas guadalupensis*; Yeo 1970; Skogerboe and Getsinger 2001), white waterlily (*Nymphaea odorata*; Skogerboe and Getsinger 2001), leafy pondweed (*Potamogeton foliosus*; Yeo 1970), Illinois pondweed (*Potamogeton illinoensis*; Skogerboe and Getsinger 2001; Shearer and Nelson 2002; Skogerboe and Getsinger 2002; Mudge 2013), long-leaf pondweed (*Potamogeton nodosus*; Yeo 1970; Skogerboe and Getsinger 2001; Shearer and Nelson 2002; Mudge 2013), small pondweed (*P. pusillus*; Yeo 1970), broadleaf arrowhead (*Sagittaria latifolia*; Skogerboe and Getsinger 2001), sago pondweed (*Stuckenia pectinata*; Yeo 1970; Sprecher et al. 1998a; Skogerboe and Getsinger 2002; Slade et al. 2008), water celery (*Vallisneria americana*; Skogerboe and Getsinger 2001; Skogerboe and Getsinger 2002; Shearer and Nelson 2002; Mudge 2013), and horned pondweed (Yeo 1970; Gyselinck and Courter 2015).

Species which were not significantly impacted or which recovered quickly include watershield (*Brasenia schreberi*; Skogerboe and Getsinger 2001), muskgrasses (*Chara* spp.; Yeo 1970; Wells and Clayton 1993; Hofstra and Clayton 2001), common waterweed (Yeo 1970; Wells and Clayton 1993; Skogerboe and Getsinger 2002), water stargrass (Skogerboe and Getsinger 2001), water net (*Hydrodictyon reticulatum*; Wells and Clayton 1993), the freshwater macroalgae *Nitella clavata* (Yeo 1970), yellow pond-lily (*Nuphar advena*; Skogerboe and Getsinger 2002), swamp smartweed (*Polygonum hydropiperoides*; Skogerboe and Getsinger 2002), pickerelweed (*Pontederia cordata*; Skogerboe and Getsinger 2001), softstem bulrush (*Schoenoplectus tabernaemontani*; Skogerboe and Getsinger 2001), and broadleaf cattail (*Typha latifolia*; Skogerboe and Getsinger 2002).

Field trials mirror the species susceptibility above and in addition show that endothall also can impact several high-value pondweed species (*Potamogeton* spp.), including large-leaf pondweed (*P. amplifolius*; Parsons et al. 2004), fern pondweed (*P. robbinsii*; Onterra 2015; Onterra 2018), white-stem pondweed (*P. praelongus*; Onterra 2018), small pondweed (Big Chetac Chain Lake Association 2016; Onterra 2018), clasping-leaf pondweed (*P. richardsonii*; Onterra 2018), and flat-stem pondweed (*P. zosteriformis*; Onterra 2017b).

Dimethylalkylamine salt formulations

The dimethylalkylamine formulations are more active on aquatic plants than the dipotassium formulations (EPA RED Endothall 2005; Keckemet 1969). At least one mesocosm study has shown that dimethylalkylamine formulation of endothall (at or below the maximum label rate) will control the invasive species fanwort (Hunt et al. 2015) and the native species common waterweed (Mudge et al. 2015), while others have shown that the dipotassium formulation does not control these species well.

Imazamox

Registration and Formulations

Imazamox is the common name of the active ingredient ammonium salt of imazamox (2-[4,5-dihydro-4-methyl-4-(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-5-(methoxymethl)-3-pyridinecarboxylic acid. It was registered with U.S. EPA in 2008 and is currently under registration review with an estimated registration decision between 2019 and 2020 (EPA Imazamox Plan 2014). In aquatic environments, a liquid formulation is typically applied to submerged vegetation by broadcast spray or underwater hose application and to emergent or floating leaf vegetation by broadcast spray or foliar application. There is also a granular formulation.

Mode of Action and Degradation

Imazamox is a systemic herbicide that moves throughout the plant tissue and prevents plants from producing a necessary enzyme, acetolactate synthase (ALS), which is not found in animals. Susceptible plants will stop growing soon after treatment, but plant death and decomposition will occur over several weeks (Mudge and Netherland 2014). If used as a post-emergence herbicide, imazamox should be applied to plants that are actively growing. Resistance to ALS-inhibiting herbicides has appeared in weeds at a higher rate than other herbicide types in terrestrial environments (Tranel and Wright 2002).

Dissipation studies in lakes indicate a half-life ranging from 4 to 49 days with an average of 17 days. Herbicide breakdown does not occur readily in deep, poorly-oxygenated water where there is no light. In this part of a lake, imazamox will tend to bind to sediments rather than breaking down, with a half-life of approximately 2 years. Once in soil, leaching to groundwater is believed to be very limited. The breakdown products of imazamox are nicotinic acid and di- and tricarboxylic acids. It has been suggested that photolytic break down of imazamox is faster than other herbicides, reducing exposure times. However, short-term imazamox exposures have also been associated with extended regrowth times relative to other herbicides (Netherland 2011).

Toxicology

Treated water may be used immediately following application for fishing, swimming, cooking, bathing, and watering livestock. If water is to be used as potable water or for irrigation, the tolerance is 0.05 ppm (50 ppb), and a 24-hour irrigation restriction may apply depending on the waterbody. None of the breakdown products are herbicidal nor suggest concerns for aquatic organisms or human health.

Most concerns about adverse effects on human health involve applicator exposure. Concentrated imazamox can cause eye and skin irritation and is harmful if inhaled. Applicators should minimize exposure by wearing long-sleeved shirts and pants, rubber gloves, and shoes and socks.

Honeybees are affected at application rates so drift during application should be minimized. Laboratory tests using rainbow trout (*Oncorhynchus mykiss*), bluegill (*Lepomis macrochirus*), and water fleas (*Daphnia magna*) indicate that imazamox is not toxic to these species at label application rates.

Imazamox is rated “practically non-toxic” to fish and aquatic invertebrates and does not bioaccumulate in fish. Additional studies on birds indicate toxicity only at dosages that exceed approved application rates.

In chronic tests, imazamox was not shown to cause tumors, birth defects or reproductive toxicity in test animals. Most studies show no evidence of mutagenicity. Imazamox is not metabolized and was excreted by mammals tested. Based on its low acute toxicity to mammals, and its rapid disappearance from the water column due to light and microbial degradation and binding to soil, imazamox is not considered to pose a risk to recreational water users.

Species Susceptibility

In Wisconsin, imazamox is used for treating non-native emergent vegetation such as non-native phragmites (*Phragmites australis* subsp. *australis*) and flowering rush (*Butomus umbellatus*). Imazamox may also be used to treat the invasive curly-leaf pondweed (*Potamogeton crispus*). Desirable native species that may be affected could include other pondweed species (long-leaf pondweed (*P. nodosus*), flat-stem pondweed (*P. zosteriformis*), leafy pondweed (*P. foliosus*), Illinois pondweed (*P. illinoensis*), small pondweed (*P. pusillus*), variable-leaf pondweed (*P. gramineus*), water-thread pondweed (*P. diversifolius*), perfoliate pondweed (*P. perfoliatus*), large-leaf pondweed (*P. amplifolius*), watershield (*Brasenia schreberi*), and some bladderworts (*Utricularia* spp.). Higher rates of imazamox will control Eurasian watermilfoil (*Myriophyllum spicatum*) but would also have greater non-target impacts on native plants. Imazamox can also be used during a drawdown to prevent plant regrowth and on emergent vegetation.

At low concentrations, imazamox can cause growth regulation rather than mortality in some plant species. This has been shown for non-native phragmites and hydrilla (*Hydrilla verticillata*; Netherland 2011; Cheshier et al. 2012; Theel et al. 2012). In the case of hydrilla, some have suggested that this effect could be used to maintain habitat complexity while providing some target species control (Theel et al. 2012). Imazamox can reduce biomass of non-native phragmites though some studies found regrowth to occur, suggesting a combination of imazapyr and glyphosate to be more effective (Cheshier et al. 2012; Knezevic et al. 2013).

Some level of control of imazamox has also been reported for water hyacinth (*Eichhornia crassipes*), parrot feather (*Myriophyllum aquaticum*), Japanese stiltgrass (*Microstegium* *vimineum*), water lettuce (*Pistia stratiotes*), and southern cattail (*Typha domingensis*; Emerine et al. 2010; de Campos et al. 2012; Rodgers and Black 2012; Hall et al. 2014; Mudge and Netherland 2014). Imazamox was observed to have greater efficacy in controlling floating plants than emergents in a study of six aquatic plant species, including water hyacinth, water lettuce, parrot feather, and giant salvinia (*Salvinia molesta*; Emerine et al. 2010). Non-target effects have been observed for softstem bulrush (*Schoenoplectus tabernaemontani*), pickerelweed (*Pontederia cordata*), and the native pondweeds long-leaf pondweed, Illinois pondweed, and coontail (*Ceratophyllum demersum*; Koschnick et al. 2007; Mudge 2013). Giant salvinia, white waterlily (*Nymphaea odorata*), bog smartweed (*Polygonum setaceum*), giant bulrush (*Schoenoplectus californicus*), water celery (*Vallisneria americana*; though the root biomass of wide-leaf *Vallisneria* may be reduced), and several algal species have been found by multiple studies to be unaffected by imazamox (Netherland et al. 2009; Emerine et al. 2010; Rodgers and Black 2012; Mudge 2013; Mudge and Netherland 2014). Other species are likely to be susceptible, for which the effects of imazamox have not yet been evaluated.

Florpyrauxifen-benzyl

Registration and Formulations

Florpyrauxifen-benzyl is a relatively new herbicide, which was first registered with the U.S. EPA in September 2017. The active ingredient is 4-amino-3-chloro-6-(4-chloro-2-fluoro-3-methoxyphenyl)-5-fluoro-pyridine-2-benzyl ester, also identified as florpyrauxifen-benzyl. Florpyrauxifen-benzyl is used for submerged, floating, and emergent aquatic plant control (e.g., ProcellaCORTM) in slow-moving and quiescent waters, as well as for broad spectrum weed control in rice (*Oryza sativa*) culture systems and other crops (e.g., RinskorTM).

Mode of Action and Degradation

Florpyrauxifen-benzyl is a member of a new class of synthetic auxins, the arylpicolinates, that differ in binding affinity compared to other currently registered synthetic auxins such as 2,4-D and triclopyr (Bell et al. 2015). Florpyrauxifen-benzyl is a systemic herbicide (Heilman et al. 2017).

Laboratory studies and preliminary field dissipation studies indicate that florpyrauxifen-benzyl in water is subject to rapid photolysis (Heilman et al. 2017). In addition, the herbicide can also convert partially via hydrolysis to an acid form at high pH (>9) and higher water temperatures (>25°C), and microbial activity in the water and sediment can also enhance degradation (Heilman et al. 2017). The acid form is noted to have reduced herbicidal activity (Netherland and Richardson 2016; Richardson et al. 2016). Under growth chamber conditions, water samples at 1 DAT found that 44-59% of the applied herbicide had converted to acid form, while sampling at 7 and 14 DAT indicated that all the herbicide had converted to acid form (Netherland and Richardson 2016). The herbicide is short-lived, with half-lives ranging from 4 to 6 days in aerobic aquatic environments, and 2 days in anaerobic aquatic environments (WSDE 2017). Degradation in surface water is accelerated when exposed to sunlight, with a reported photolytic half-life in laboratory testing of 0.07 days (WSDE 2017).

There is some anecdotal evidence that initial water temperature and/or pH may impact the efficacy of florpyrauxifen-benzyl (Beets and Netherland 2018). Florpyrauxifen-benzyl has a high soil adsorption coefficient (KOC) and low volatility, which allows for rapid plant uptake resulting in short exposure time requirements (Heilman et al. 2017). Florpyrauxifen-benzyl degrades quickly (2-15 days) in soil and sediment (Netherland et al. 2016). Few studies have yet been completed for groundwater, but based on known environmental properties, florpyrauxifen-benzyl is not expected to be associated with potential environmental impacts in groundwater (WSDE 2017).

Toxicology

No adverse human health effects were observed in toxicological studies submitted for EPA herbicide registration, regardless of the route of exposure (Heilman et al. 2017). There are no drinking water or recreational use restrictions, including swimming and fishing. There are no restrictions on irrigating turf, and a short waiting period (dependent on application rate) for other non-agricultural irrigation purposes.

Florpyrauxifen-benzyl showed a good environmental profile for use in water, and is "practically non-toxic" to birds, bees, reptiles, amphibians, and mammals (Heilman et al. 2017). No ecotoxicological effects were observed on freshwater mussel or juvenile chinook salmon (Heilman et al. 2017). Florpyrauxifen-benzyl will temporarily bioaccumulate in freshwater organisms but is rapidly depurated and/or metabolized within 1 to 3 days after exposure to high (>150 ppb) concentrations (WSDE 2017).

An LC50 value indicates the concentration of a chemical required to kill 50% of a test population of organisms. LC50 values are commonly used to describe the toxicity of a substance. Label recommendations for milfoils do not exceed 9.65 ppb and the maximum label rate for an acre-foot of water is 48.25 ppb. Acute toxicity results using rainbow trout (*Oncorhynchus mykiss*), fathead minnow (*Pimephales promelas*), and sheepshead minnows (*Cyprinodon variegatus variegatus*) indicated LC50 values of greater than 49 ppb, 41 ppb, and 40 ppb, respectively when exposed to the technical grade active ingredient (WSDE 2017). An LC50 value of greater than 1,900 ppb was reported for common carp (*Cyprinus carpio*) exposed to the ProcellaCOR end-use formulation (WSDE 2017).

Acute toxicity results for the technical grade active ingredient using water flea (*Daphnia magna*) and midge (*Chironomus* sp.) indicated LC50 values of greater than 62 ppb and 60 ppb, respectively (WSDE 2017). Comparable acute ecotoxicity testing performed on *D. magna* using the ProcellaCOR end-use formulation indicated an LC50 value of greater than 8 ppm (80,000 ppb; WSDE 2017).

The ecotoxicological no observed effect concentration (NOEC) for various organisms as reported by Netherland et al. (2016) are: fish (>515 ppb ai), water flea (*Daphnia* spp.; >21440 ppb ai), freshwater mussels (>1023 ppb ai), saltwater mysid (>362 ppb ai), saltwater oyster (>289 ppb ai), and green algae (>480 ppb ai). Additional details on currently available ecotoxicological information is compiled by WSDE (2017).

Species Susceptibility

Florpyrauxifen-benzyl is a labeled for control of invasive watermilfoils (e.g., Eurasian watermilfoil (*Myriophyllum spicatum*), hybrid watermilfoil (*M. spicatum* x *sibiricum*), parrot feather (*M. aquaticum*)), hydrilla (*Hydrilla verticillata*), and other non-native floating plants such as floating hearts (*Nymphoides* spp.), water hyacinth (*Eichhornia crassipes*), and water chestnut (*Trapa natans*; Netherland and Richardson 2016; Richardson et al. 2016). Natives species listed on the product label as susceptible to florpyrauxifen-benzyl include coontail (*Ceratophyllum demersum*; Heilman et al. 2017), watershield (*Brasenia schreberi*), and American lotus (*Nelumbo lutea*). In laboratory settings, pickerelweed (*Pontederia cordata*) vegetation has also been shown to be affected (Beets and Netherland 2018).

Based on available data, florpyrauxifen-benzyl appears to show few impacts to native aquatic plants such as aquatic grasses, bulrush (*Schoenoplectus* spp.), cattail (*Typha* spp.), pondweeds (*Potamogeton* spp.), naiads (*Najas* spp.), and water celery (*Vallisneria americana*; WSDE 2017). Laboratory and mesocosm studies also found water marigold (*Bidens beckii*), white waterlily (*Nymphaea odorata*), common waterweed (*Elodea canadensis*), water stargrass (*Heteranthera dubia*), long-leaf pondweed (*Potamogeton nodosus*), and Illinois pondweed (*P. illinoensis*) to be relatively less sensitive to florpyrauxifen-benzyl than labeled species (Netherland et al. 2016; Netherland and Richardson 2016). Non-native fanwort (*Cabomba caroliniana*) was also found to be tolerant in laboratory study (Richardson et al. 2016).

Since florpyrauxifen-benzyl is a relatively new approved herbicide, detailed information on field applications is very limited. Trials in small waterbodies have shown control of parrot feather (*Myriophyllum aquaticum*), variable-leaf watermilfoil (*M. heterophyllum*), and yellow floating heart (*Nymphoides peltata*; Heilman et al. 2017).

S.3.3.3. Emergent and Wetland Herbicides

Glyphosate

Registration and Formulations

Glyphosate is a commonly used herbicide that is utilized in both aquatic and terrestrial sites. It was first registered for use in 1974. EPA is currently re-evaluating glyphosate and the registration decision was expected in 2014 (EPA Glyphosate Plan 2009). The use of glyphosate-based herbicides in aquatic environments that are not approved for aquatic use is very unsafe and is a violation of federal and state pesticide laws. Different formulations of glyphosate are available, including isopropylamine salt of glyphosate and potassium glyphosate.

Glyphosate is effective only on plants that grow above the water and needs to be applied to plants that are actively growing. It will not be effective on plants that are submerged or have most of their foliage underwater, nor will it control regrowth from seed.

Mode of Action and Degradation

Glyphosate is a systemic herbicide that moves throughout the plant tissue and works by inhibiting an important enzyme needed for multiple plant processes, including growth. Following treatment, plants will gradually wilt, appear yellow, and will die in approximately 2 to 7 days. It may take up to 30 days for these effects to become apparent for woody species.

Application should be avoided when heavy rain is predicted within 6 hours. To avoid drift, application is not recommended when winds exceed 5 mph. In addition, excessive speed or pressure during application may allow spray to drift and must be avoided. Effectiveness of glyphosate treatments may be reduced if applied when plants are growing poorly, such as due to drought stress, disease, or insect damage. A surfactant approved for aquatic sites must be mixed with glyphosate before application.

In water, the concentration of glyphosate is reduced through dispersal by water movement, binding to the sediments, and break-down by microorganisms. The half-life of glyphosate is between 3 and 133 days, depending on water conditions. Glyphosate disperses rapidly in water so dilution occurs quickly, thus moving water will decrease concentration, but not half-life. The primary breakdown product of glyphosate is aminomethylphosphonic acid (AMPA), which is also degraded by microbes in water and soil.

Toxicology

Most aquatic forms of glyphosate have no restrictions on swimming or eating fish from treated waterbodies. However, potable water intakes within ½ mile of application must be turned off for 48 hours after treatment. Different formulations and products containing glyphosate may vary in post-treatment water use restrictions.

Most glyphosate-related health concerns for humans involve applicator exposure, exposure through drift, and the surfactant exposure. Some adverse effects from direct contact with the herbicide include temporary symptoms of dermatitis, eye ailments, headaches, dizziness, and nausea. Protective clothing (goggles, a face shield, chemical resistant gloves, aprons, and footwear) should be worn by applicators to reduce exposure. Recently it has been demonstrated that terrestrial formulations of glyphosate can have toxic effects to human embryonic cells and linked to endocrine disruption (Benachour et al. 2007; Gasnier et al. 2009).

Laboratory testing indicates that glyphosate is toxic to carp (*Cyprinus* spp.), bluegills (*Lepomis macrochirus*), rainbow trout (*Oncorhynchus mykiss*), and water fleas (*Daphnia* spp.) only at dosages well above the label application rates. Similarly, it is rated "practically non-toxic" to other aquatic species tested. Studies by other researchers examining the effects of glyphosate on important food chain organisms such as midge larvae, mayfly nymphs, and scuds have demonstrated a wide margin of safety between application rates.

EPA data suggest that toxicological effects of the AMPA compound are similar to that of glyphosate itself. Glyphosate also contains a nitrosamine (n-nitroso-glyphosate) as a contaminant at levels of 0.1 ppm or less. Tests to determine the potential health risks of nitrosamines are not required by the EPA unless the level exceeds 1.0 ppm.

Species Susceptibility

Glyphosate is only effective on actively growing plants that grow above the water's surface. It can be used to control reed canary grass (*Phalaris arundinacea*), cattails (*Typha* spp.; Linz et al. 1992; Messersmith et al. 1992), purple loosestrife (*Lythrum salicaria*), phragmites (*Phragmites australis* subsp. *australis*; Back and Holomuzki 2008; True et al. 2010; Back et al. 2012; Cheshier et al. 2012), water hyacinth (*Eichhornia crassipes*; Lopez 1993; Jadhav et al. 2008), water lettuce (*Pistia stratiotes*; Mudge and Netherland 2014), water chestnut (*Trapa natans*; Rector et al. 2015), Japanese stiltgrass (*Microstegium vimineum*; Hall et al. 2014), giant reed (*Arundo donax*; Spencer 2014), and perennial pepperweed (*Lepidium latifolium*; Boyer and Burdick 2010). Glyphosate will also reduce abundance of white waterlily (*Nymphaea odorata*) and pond-lilies (*Nuphar* spp.; Riemer and Welker 1974). Purple loosestrife biocontrol beetle (*Galerucella calmariensis*) oviposition and survival have been shown not to be affected by integrated management with glyphosate. Studies have found pickerelweed (*Pontederia cordata*) and floating marsh pennywort (*Hydrocotyle ranunculoides*) to be somewhat tolerant to glyphosate (Newman and Dawson 1999; Gettys and Sutton 2004).

Imazapyr

Registration and Formulations

Imazapyr was registered with the U.S. EPA for aquatic use in 2003 and is currently under registration review. It was estimated to have a registration review decision in 2017 (EPA Imazapyr Plan 2014). The active ingredient is isopropylamine salt of imazapyr (2-[4,5-dihydro-4-methyl-4-(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-3-pyridinecarboxylic acid). Imazapyr is used for control of emergent and floating-leaf vegetation. It is not recommended for control of submersed vegetation.

Mode of Action and Degradation

Imazapyr is a systemic herbicide that moves throughout the plant tissue and prevents plants from producing a necessary enzyme, acetolactate synthase (ALS), which is not found in animals. Susceptible plants will stop growing soon after treatment and become reddish at the tips of the plant. Plant death and decomposition will occur gradually over several weeks to months. Imazapyr should be applied to plants that are actively growing. If applied to mature plants, a higher concentration of herbicide and a longer contact time will be required.

Imazapyr is broken down in the water by light and has a half-life ranging from three to five days. Three degradation products are created as imazapyr breaks down: pyridine hydroxy-dicarboxylic acid, pyridine dicarboxylic acid (quinolinic acid), and nicotinic acid. These degradates persist in water for approximately the same amount of time as imazapyr (half-lives of three to eight days). In soils imazapyr is broken down by microbes, rather than light, and persists with a half-life of one to five months (Boyer and Burdick 2010). Imazapyr doesn't bind to sediments, so leaching through soil into groundwater is likely.

Toxicology

There are no restrictions on recreational use of treated water, including swimming and eating fish from treated waterbodies. If application occurs within a ½ mile of a drinking water intake, then the intake must be shut off for 48 hours following treatment. There is a 120-day irrigation restriction for treated water, but irrigation can begin sooner if the concentration falls below 0.001 ppm (1 ppb). Imazapyr degradates are no more toxic than imazapyr itself and are excreted faster than imazapyr when ingested.

Concentrated imazapyr has low acute toxicity on the skin or if ingested but is harmful if inhaled and may cause irreversible damage if it gets in the eyes. Applicators should wear chemical-resistant gloves while handling, and persons not involved in application should avoid the treatment area during treatment. Chronic toxicity tests for imazapyr indicate that it is not carcinogenic, mutagenic, or neurotoxic. It also does not cause reproductive or developmental toxicity and is not a suspected endocrine disrupter.

Imazapyr is "practically non-toxic" to fish, invertebrates, birds and mammals. Studies have also shown imazapyr to be "practically non-toxic" to "slightly toxic" to tadpoles and juvenile frogs (Trumbo and Waligora 2009; Yahnke et al. 2013). Toxicity tests have not been published on reptiles. Imazapyr does not bioaccumulate in animal tissues.

Species Susceptibility

The imazapyr herbicide label is listed to control the invasive plants phragmites (*Phragmites australis* subsp. *australis*), purple loosestrife (*Lythrum salicaria*), reed canary grass (*Phalaris arundinacea*), non-native cattails (*Typha* spp.) and Japanese knotweed (*Fallopia japonica*) in Wisconsin. Native species that are also controlled include cattails (*Typha* spp.), waterlilies (*Nymphaea* sp.), pickerelweed (*Pontederia cordata*), duckweeds (*Lemna* spp.), and arrowhead (*Sagittaria* spp.).

Studies have shown imazapyr to effectively control giant reed (*Arundo donax*), water hyacinth (*Eichhornia crassipes*), manyflower marsh-pennywort (*Hydrocotyle umbellata*); yellow iris (*Iris pseudacorus*), water lettuce (*Pistia stratiotes*), perennial pepperweed (*Lepidium latifolium*), Japanese stiltgrass (*Microstegium vimineum*), parrot feather (*Myriophyllum aquaticum*), and cattails (Boyer and Burdick 2010; True et al. 2010; Back et al. 2012; Cheshier et al. 2012; Whitcraft and Grewell 2012; Hall et al. 2014; Spencer 2014; Cruz et al. 2015; DiTomaso and Kyser 2016). Giant salvinia (*Salvinia molesta*) was found to be imazapyr-tolerant (Nelson et al. 2001).

S.3.3.4. Herbicides Used for Submersed and Emergent Plants

Triclopyr

Registration and Formulations

Triclopyr was initially registered with the U.S. EPA in 1979, reregistered in 1997, and is currently under review with an estimated registration review decision in 2019 (EPA Triclopyr Plan 2014). There are two forms of triclopyr used commercially as herbicides: the triethylamine salt (TEA) and the butoxyethyl ester (BEE). BEE formulations are considered highly toxic to aquatic organisms, with observed lethal effects on fish (Kreutzweiser et al. 1994) as well as avoidance behavior and growth impairment in amphibians (Wojtaszek et al. 2005). The active ingredient triethylamine salt (3,5,6-trichloro-2-pyridinyloxyacetic acid) is the formulation registered for use in aquatic systems. It is sold both in liquid and granular forms for control of submerged, emergent, and floating-leaf vegetation. There is also a liquid premixed formulation that contains triclopyr and 2,4-D, which when combined together are reported to have synergistic impacts. Only triclopyr products labeled for use in aquatic environments may be used to control aquatic plants.

Mode of Action and Degradation

Triclopyr is a systemic plant growth regulator that is believed to selectively act on broadleaf (dicot) and woody plants. Following treatment, triclopyr is taken up through the roots, stems and leaf tissues, plant growth becomes abnormal and twisted, and plants die within one to two weeks after application (Getsinger et al. 2000). Triclopyr is somewhat persistent and can move through soil, although only mobile enough to permeate top soil layers and likely not mobile enough to potentially contaminate groundwater (Lee et al. 1986; Morris et al. 1987; Stephenson et al. 1990).

Triclopyr is broken down rapidly by light (photolysis) and microbes, while hydrolysis is not a significant route of degradation. Triclopyr photodegrades and is further metabolized to carbon dioxide, water, and various organic acids by aquatic organisms (McCall and Gavit 1986). It has been hypothesized that the major mechanism for the removal of triclopyr from the aquatic environment is microbial degradation, though the role of photolysis likely remains important in near-surface and shallow waters (Petty et al. 2001). Degradation of triclopyr by microbial action is slowed in the absence of light (Petty et al. 2003). Triclopyr is very slowly degraded under anaerobic conditions, with a reported half-life (the time it takes for half of the active ingredient to degrade) of about 3.5 years (Laskowski and Bidlack 1984). Another study of triclopyr under aerobic aquatic conditions yielded a half-life of 4.7 months (Woodburn and Cranor 1987). The initial breakdown products of triclopyr are TCP (3,5,6-trichloro-2-pyridinol) and TMP (3,5,6-trichloro-2-methoxypridine).

Several studies reported triclopyr half-lives between 0.5-7.5 days (Woodburn et al. 1993; Getsinger et al. 2000; Petty et al. 2001; Petty et al. 2003). Two large-scale, low-dose treatments were reported to have longer triclopyr half-lives from 3.7-12.1 days (Netherland and Jones 2015). Triclopyr half-lives have been shown to range from 3.4 days in plants, 2.8-5.8 days in sediment, up to 11 days in fish tissue, and 11.5 days in crayfish (Woodburn et al. 1993; Getsinger et al. 2000; Petty et al. 2003). TMP and TCP may have longer half-lives than triclopyr, with higher levels in bottom-feeding fish and the inedible parts of fish (Getsinger et al. 2000).

Toxicology

Based upon the triclopyr herbicide label, there are no restrictions on swimming, eating fish from treated waterbodies, or pet/livestock drinking water use. Before treated water can be used for irrigation, the concentration must be below 0.001 ppm (1 ppb), or at least 120 days must pass. Treated water should not be used for drinking water until concentrations of triclopyr are less than

0.4 ppm (400 ppb). There is a least one case of direct human ingestion of triclopyr TEA which resulted in metabolic acidosis and coma with cardiovascular impairment (Kyong et al. 2010).

There are substantial differences in toxicity of BEE and TEA, with the BEE shown to be more toxic in aquatic settings. BEE formulations are considered highly toxic to aquatic organisms, with observed lethal effects on fish (Kreutzweiser et al. 1994) as well as avoidance behavior and growth impairment in amphibians (Wojtaszek et al. 2005). Triclopyr TEA is "practically non-toxic" to freshwater fish and invertebrates (Mayes et al. 1984; Gersich et al. 1984). It ranges from "practically non-toxic" to "slightly toxic" to birds (EPA Triclopyr RED 1998). TCP and TMP appear to be slightly more toxic to aquatic organisms than triclopyr; however, the peak concentration of these degradates is low following treatment and depurates from organisms readily, so that they are not believed to pose a concern to aquatic organisms.

Species susceptibility

Triclopyr has been used to control Eurasian watermilfoil (*Myriophyllum spicatum*) and hybrid watermilfoil (*M. spicatum* x *sibiricum*) at both small- and large-scales (Netherland and Getsinger 1992; Getsinger et al. 1997; Poovey et al. 2004; Poovey et al. 2007; Nelson and Shearer 2008; Heilman et al. 2009; Glomski and Netherland 2010; Netherland and Glomski 2014; Netherland and Jones 2015). Getsinger et al. (2000) found that peak triclopyr accumulation was higher in Eurasian watermilfoil than flat-stem pondweed (*Potamogeton zosteriformis*), indicating triclopyr's affinity for Eurasian watermilfoil as a target species.

According to product labels, triclopyr is capable of controlling or affecting many emergent woody plant species, purple loosestrife (*Lythrum salicaria*), phragmites (*Phragmites australis* subsp. *australis*), American lotus (*Nelumbo lutea*), milfoils (*Myriophyllum* spp.), and many others. Triclopyr application has resulted in reduced frequency of occurrence, reduced biomass, or growth regulation for the following species: common waterweed (*Elodea canadensis*), water stargrass (*Heteranthera dubia*), white waterlily (*Nymphaea odorata*), purple loosestrife, Eurasian watermilfoil, parrot feather (*Myriophyllum aquaticum*), variable-leaf watermilfoil (*M. heterophyllum*), watercress (*Nasturtium officinale*), phragmites, flat-stem pondweed (*Potamogeton zosteriformis*), clasping-leaf pondweed (*P. richardsonii*), stiff pondweed (*P. strictifolius*), variable-leaf pondweed (*P. gramineus*), white water crowfoot (*Ranunculus aquatilis*), sago pondweed (*Stuckenia pectinata*), softstem bulrush (*Schoenoplectus tabernaemontani*), hardstem bulrush (*S. acutus*), water chestnut (*Trapa natans*), duckweeds (*Lemna* spp.), and submerged flowering rush (*Butomus umbellatus*; Cowgill et al. 1989; Gabor et al. 1995; Sprecher and Stewart 1995; Getsinger et al. 2003; Poovey et al. 2004; Hofstra et al. 2006; Poovey and Getsinger 2007; Champion et al. 2008; Derr 2008; Glomski and Nelson 2008; Glomski et al. 2009; True et al. 2010; Cheshier et al. 2012; Netherland and Jones 2015; Madsen et al. 2015; Madsen et al. 2016). Wild rice (*Zizania palustris*) biomass and height has been shown to decrease significantly following triclopyr application at 2.5 mg/L. Declines were not significant at lower concentrations (0.75 mg/L), though seedlings were more sensitive than young or mature plants (Madsen et al. 2008). American bulrush (*Schoenoplectus americanus*), spatterdock (*Nuphar variegata*), fern pondweed (*Potamogeton robbinsii*), large-leaf pondweed (*P. amplifolius*), leafy pondweed (*P. foliosus*), white-stem pondweed (*P. praelongus*), long-leaf pondweed (*P. nodosus*), Illinois pondweed (*P. illinoensis*), and water celery (*Vallisneria americana*) can be somewhat tolerant of triclopyr applications depending on waterbody characteristics and application rates (Sprecher and Stewart 1995; Glomski et al. 2009; Wersal et al. 2010b; Netherland and Glomski 2014).

Netherland and Jones (2015) evaluated the impact of large-scale, low-dose (~0.1-0.3 ppm) granular triclopyr) applications for control of non-native watermilfoil on several bays of Lake Minnetonka, Minnesota. Near complete loss of milfoil in the treated bays was observed the year of treatment, with increased milfoil frequency reported the following season. However, despite the observed increase in frequency, milfoil biomass remained a minor component of bay-wide biomass (<2%). The number of points with native plants, mean native species per point, and native species richness in the bays were not reduced following treatment. However, reductions in frequency were seen amongst individual species, including northern watermilfoil (*Myriophyllum sibiricum*), water stargrass, common waterweed, and flat-stem pondweed.

Penoxsulam

Registration and Formulations

Penoxsulam (2-(2,2-difluoroethoxy)--6-(trifluoromethyl-N-(5,8-dimethoxy[1,2,4] triazolo[1,5-c]pyrimidin-2-yl))benzenesulfonamide), also referred to as DE-638, XDE-638, XR-638 is a post-emergence, acetolactate synthase (ALS) inhibiting herbicide. It was first registered for use by the U.S. EPA in 2009. It is liquid in formulation and used for large-scale control of submerged, emergent, and floating-leaf vegetation. Information presented here can be found in the EPA pesticide fact sheet (EPA Penoxsulam 2004).

Mode of Action and Degradation

Penoxsulam is a slow-acting herbicide that is absorbed by above- and below-ground plant tissue and translocated throughout the plant. Penoxsulam interferes with plant growth by inhibiting the AHAS/ALS enzyme which in turn inhibits the production of important amino acids (Tranel and Wright 2002). Plant injury or death usually occurs between 2 and 4 weeks following application.

Penoxsulam is highly mobile but not persistent in either aquatic or terrestrial settings. However, the degradation process is complex. Two degradation pathways have been identified that result in at least 13 degradation products that persist for far longer than the original chemical. Both microbial- and photo-degradation are likely important means by which the herbicide is removed from the environment (Monika et al. 2017). It is relatively stable in water alone without sunlight, which means it may persist in light-limited areas.

The half-life for penoxsulam is between 12 and 38 days. Penoxsulam must remain in contact with plants for around 60 days. Thus, supplemental applications following initial treatment may be required to maintain adequate concentration exposure time (CET). Due to the long CET requirement, penoxsulam is likely best suited to large-scale or whole-lake applications.

Toxicology

Penoxsulam is unlikely to be toxic to animals but may be "slightly toxic" to birds that consume it. Human health studies have not revealed evidence of acute or chronic toxicity, though some indication of endocrine disruption deserves further study. However, screening-level assessments of risk have not been conducted on the major degradates which may have unknown non-target effects. Penoxsulam itself is unlikely to bioaccumulate in fish.

Species Susceptibility

Penoxsulam is used to control monocot and dicot plant species in aquatic and terrestrial environments. The herbicide is often applied at low concentrations of 0.002-0.02 ppm (2-20 ppb), but as a result long exposure times are usually required for effective target species control (Cheshier et al. 2011; Mudge et al. 2012b). For aquatic plant management applications, penoxsulam is most commonly utilized for control of hydrilla (*Hydrilla verticillata*). It has also been used for control of giant salvinia (*Salvinia molesta*), water hyacinth (*Eichhornia crassipes*), and water lettuce (*Pistia stratiotes*; Richardson and Gardner 2007; Mudge and Netherland 2014). However, the herbicide is only semi-selective; it has been implicated in injury to non-target emergent native species, including arrowheads (*Sagittaria* spp.) and spikerushes (*Eleocharis* spp.) and free-floating species like duckweed (Mudge and Netherland 2014; Cheshier et al. 2011). Penoxsulam can also be used to control milfoils such as Eurasian watermilfoil (*Myriophyllum spicatum*) and variable-leaf watermilfoil (*M. heterophyllum*; Glomski and Netherland 2008). Seedling emergence as well as vegetative vigor is impaired by penoxsulam in both dicots and monocots, so buffer zone and dissipation reduction strategies may be necessary to avoid non-target impacts (EPA Penoxsulam 2004).

When used to treat salvinia, the herbicide was found to have effects lasting through 10 weeks following treatment (Mudge et al. 2012b). The herbicide is effective at low doses, but while low-concentration applications of slow-acting herbicides like penoxsulam often result in temporary growth regulation and stunting, plants are likely to recover following treatment. Thus, complementary management strategies should be employed to discourage early regrowth (Mudge et al. 2012b). In particular, joint biological and herbicidal control with penoxsulam has shown good control of water hyacinth (Moran 2012). Alternately, a low concentration may be maintained over time by repeated low-dose applications. Studies show that maintaining a low concentration for at least 8-12 weeks provided excellent control of salvinia, and that a low dose followed by a high-dose application was even more efficacious (Mudge et al. 2012b).

S.3.4. Physical Removal Techniques

There are several management options which involve physical removal of aquatic plants, either by manual or mechanical means. Some of these include manual and mechanical cutting and hand-pulling or Diver-Assisted Suction Harvesting (DASH).

S.3.4.1. Manual and Mechanical Cutting

Manual and Mechanical Cutting

Manual and mechanical cutting involve slicing off a portion of the target plants and removing the cut portion from the waterbody. In addition to actively removing parts of the target plants, destruction of vegetative material may help prevent further plant growth by decreasing photosynthetic uptake, and preventing the formation of rhizomes, tubers, and other growth types (Dall Armellina et al. 1996a, 1996b; Fox et al. 2002). These approaches can be quick to allow recreational use of a waterbody but because the plant is still established and will continue to grow from where it was cut, it often serves to provide short-term relief (Bickel and Closs 2009; Crowell et al. 1994). A synthesis of numerous historical mechanical harvesting studies is compiled by Breck et al. 1979.

The amount of time for macrophytes to return to pre-cutting levels can vary between waterbodies and with the dominant plant species present (Kaenel et al. 1998). Some studies have suggested that annual or biannual cutting of Eurasian watermilfoil (*Myriophyllum spicatum*) may be needed, while others have shown biomass can remain low the year after cutting (Kimbel and Carpenter 1981; Painter 1988; Barton et al. 2013). Hydrilla (*Hydrilla verticillata*) has been shown to recover beyond pre-harvest levels within weeks in some cases (Serafy et al. 1994). In deeper waters, greater cutting depth may lead to increased persistence of vegetative control (Unmuth et al. 1998; Barton et al. 2013). Higher frequency of cutting, rather than the amount of plant that is cut, can result in larger reductions to propagules such as turions (Fox et al. 2002).

The timing of cutting operations, as for other management approaches, is important. For species dependent on vegetative propagules, control methods should be taken before the propagules are formed. However, for species with rhizomes, cutting too early in the season merely postpones growth while later-season cutting can better reduce plant abundance (Dall Armellina et al. 1996a, 1996b). Eurasian watermilfoil regrowth may be slower if cutting is conducted later in the summer (June or later). Cutting in the fall, rather than spring or summer, may result in the lowest amount of Eurasian watermilfoil regrowth the year after management (Kimbel and Carpenter 1981). However, managing early in the growing season may reduce non-target impacts to native plant populations when early-growing non-native plants are the dominant targets (Nichols and Shaw 1986). Depending on regrowth rate and management goals, multiple harvests per growing season may be necessary (Rawls 1975).

Vegetative fragments which are not collected after cutting can produce new localized populations, potentially leading to higher plant densities (Dall Armellina et al. 1996a). Eurasian watermilfoil and common waterweed (*Elodea canadensis*) biomass can be reduced by cutting (Abernethy et al. 1996), though Eurasian watermilfoil can maintain its growth rate following cutting by developing a more-densely branched form (Rawls 1975; Mony et al. 2011). Cutting and physical removal tend to be less expensive but require more effort than benthic barriers, so these approaches may be best used for small infestations or where non-native and native species inhabit the same stand (Bailey and Calhoun 2008).

<u>Ecological Impacts of Manual and Mechanical Cutting</u>

Plants accrue nutrients into their tissues, and thus plant removal may also remove nutrients from waterbodies (Boyd 1970), though this nutrient removal may not be significant among all lake types. Cutting and harvesting of aquatic plants can lead to declines in fish as well as beneficial zooplankton, macroinvertebrate, and native plant and mussel populations (Garner et al. 1996; Aldridge 2000; Torn et al. 2010; Barton et al. 2013). Many studies suggest leaving some vegetated areas undisturbed to reduce negative effects of cutting on fish and other aquatic organisms (Swales 1982; Garner et al. 1996; Unmuth et al. 1998; Aldridge 2000; Greer et al. 2012). Recovery of these populations to cutting in the long-term is understudied and poorly understood (Barton et al. 2013). Effects on water quality can be minimal but nutrient cycling may be affected in wetland systems (Dall Armellina et al. 1996a; Martin et al. 2003). Cutting can also increase algal production, and turbidity temporarily if sediments are disturbed (Wile 1978; Bailey and Calhoun 2008).

Some changes to macroinvertebrate community composition can occur as a result of cutting (Monahan and Caffrey 1996; Bickel and Closs 2009). Studies have also shown 12-85% reductions in macroinvertebrates following cutting operations in flowing systems (Dawson et al. 1991; Kaenel et al. 1998). Macroinvertebrate communities may not rebound to pre-management levels for 4-6 months and species dependent on aquatic plants as habitat (such as simuliids and chironomids) are likely to be most affected. Reserving cutting operations for summer, rather than spring, may reduce impacts to macroinvertebrate communities (Kaenel et al. 1998).

Mechanical harvesting can also incidentally remove fish and turtles inhabiting the vegetation and lead to shifts in aquatic plant community composition (Engel 1990; Booms 1999). Studies have shown mechanical harvesting can remove between 2%-32% of the fish community by fish number, with juvenile game fish and smaller species being the primary species removed (Haller et al. 1980; Mikol 1985). Haller et al. (1980) estimated a 32% reduction in the fish community at a value of $6000/hectare. However, fish numbers rebounded to similar levels as an unmanaged area within 43 days after harvesting in the Potomac River in Maryland (Serafy et al. 1994). In addition to direct impacts to fish populations, reductions in fish growth rates may correspond with declines in zooplankton populations in response to cutting (Garner et al. 1996).

S.3.4.2. Hand Pulling and Diver-Assisted Suction Harvesting

Hand-pulling and DASH involve removing rooted plants from the bottom sediment of the water body. The entire plant is removed and disposed of elsewhere. Hand-pulling can be done at shallower depths whereas DASH, in which SCUBA divers do the pulling, may be better suited for deeper aquatic plant beds. As a permit condition, DASH and hand-pulling may not result in lifting or removal of bottom sediment (i.e., dredging). Efforts should be made to preserve water clarity because turbid conditions reduce visibility for divers, slowing the removal process and making species identification difficult. When operated with the intent to distinguish between species and minimize disturbance to desirable vegetation, DASH can be selective and provide multi-year control (Boylen et al. 1996). One study found reduced cover of Eurasian watermilfoil both in the year of harvest and the following year, along with increased native plant diversity and reduced overall plant cover the year following DASH implementation (Eichler et al. 1993). However, hand harvesting or DASH may require a large time or economic investment for Eurasian watermilfoil and other aquatic vegetation control on a large-scale (Madsen et al. 1989; Kelting and Laxson 2010). Lake type, water clarity, sediment composition, underwater obstacles and presences of dense native plants, may slow DASH efforts or even prohibit the ability to utilized DASH. Costs of DASH per acre have been reported to typically range from approximately $5,060-8,100 (Cooke et al. 1993; Mattson et al. 2004). Additionally, physical removal of turions from sediments, when applicable, has been shown to greatly reduce plant abundance for multiple subsequent growing seasons (Caffrey and Monahan 2006), though this has not been implemented in Wisconsin due to the significant effort it requires.

Ecological Impacts of Hand-Pulling and DASH

Because divers are physically uprooting plants from the lake bed, hand removal may disturb benthic organisms. Additionally, DASH may also result in some accidental capture of fish and invertebrates, small amounts of sediment removal, or increased turbidity. It is possible that equipment modifications could help minimize some of these unintended effects. Because DASH is a relatively new management approach, less information is available about potential impacts than for some more established techniques like large-scale mechanical harvesting.

S.3.4.3. Benthic Barriers

Benthic barriers can be used to kill existing plants or prevent their growth from the outset. They are sometimes referred to as benthic mats, or screens, and involve placing some sort of covering over a plant bed, which provides a physical obstruction to plant growth and reduces light availability. They may be best used for dense, confined infestations or along shore or for providing boat lanes (Engel 1983; Payne et al. 1993; Bailey and Calhoun 2008). Reductions in abundance of live aquatic plants beneath the barrier may be seen within weeks (Payne et al. 1993; Carter et al. 1994). The target plant species, light availability, and sediment accumulation have been shown to influence the efficacy of benthic barriers for aquatic plant control. Effects on the target plants may be more rapid in finer sediments because anoxic conditions are reached more quickly due to higher sediment organic content and oxidization by bacteria (Carter et al. 1994). Benthic barriers may be more expensive but less time intensive than some of the physical removal approaches described above (Carter et al. 1994; Bailey and Calhoun 2008). Engel (1983) suggests that benthic barriers may be useful in situations where plants are growing too deep for other physical removal approaches or effective herbicide application. They may also improve plant control when used in combination with herbicide treatments to hold most of the herbicide to a given treatment area (Helsel et al. 1996).

There is some necessary upkeep associated with the use of benthic barriers. Some barriers can be difficult to re-use because of algae and plants that can grow on top of the barrier. Periodically removing sediment that accumulates on the barrier can help offset this (Engel 1983; Carter et al. 1994; Laitala et al. 2012). Some materials are made to be removed after the growing season, which may make cleaning and re-use easier (Engel 1983). Additionally, gases often accumulate beneath benthic barriers as a result of plant decay, which can cause them to rise off the bottom of the waterbody, requiring further maintenance (Engel 1983; Ussery et al. 1997; Bailey and Calhoun 2008). Eurasian watermilfoil (*Myriophyllum spicatum*) and other plant species have been shown to recolonize the managed area quickly following barrier removal (Eichler et al. 1995; Boylen et al. 1996), so this approach may require hand-pulling or other integrated approaches once the barrier is removed (Carter et al. 1994; Eichler et al. 1995; Bailey and Calhoun 2008). Some studies have observed low abundance of plants maintained for 1-2 months after barriers were removed (Engel 1983). Others found that combining 2,4-D treatments with benthic barriers could reduce Eurasian watermilfoil to a degree that helped native plants recolonize the target site (Helsel et al. 1996).

The material used to create benthic barriers can vary and include biodegradable jute matting, fiberglass screens, and woven polypropylene fibers (Mayer 1978; Perkins et al. 1980; Lewis et al. 1983; Hoffman et al. 2013). Some plants such as Eurasian watermilfoil and common waterweed (Elodea canadensis; Eichler et al. 1995) are able to growth through the mesh in woven barriers but this material can be effective in reducing growth on certain target plant species (Payne et al. 1993; Caffrey et al. 2010; Hoffman et al. 2013). Hofstra and Clayton (2012) suggested that less dense materials barriers may provide selective control of some species while allowing more tolerant species, such as some charophytes (*Chara* spp. and *Nitella* spp.), to grow through. More dense materials may prevent growth of a wider range of aquatic plants (Hofstra and Clayton 2012). Most materials must be well anchored to the bottom of the waterbody, which can be accomplished early in the growing season or by placing the barriers on ice before thawing of the waterbody (Engel 1983). Gas accumulation can occur in using both fibrous mesh and screen-type barriers (Engel 1983).

Eurasian watermilfoil and common waterweed have been found to be somewhat resistant to control by benthic barriers (Perkins et al. 1980; Engel 1983) while affected species include hydrilla (*Hydrilla verticillata*), curly-leaf pondweed (*Potamogeton crispus*), and coontails (*Ceratophyllum* spp.; Engel 1983; Payne et al. 1993; Carter et al. 1994). One study found that an 8-week barrier placement removed Eurasian watermilfoil while allowing native plant regrowth after the barrier was retrieved; while shorter durations were less effective in reducing Eurasian watermilfoil abundance and longer durations negatively impacted native plant regrowth (Laitala et al. 2012).

<u>Ecological Impacts of Benthic Barriers</u>

Macroinvertebrates will be negatively affected by benthic barriers while they are in place (Engel 1983) but have been shown to rebound to pre-management conditions shortly after removal of the barrier (Payne et al. 1993; Ussery et al. 1997). Benthic barriers may also affect spawning of some warm water fish species through direct disruption of spawning habitat (NYSFOLA 2009). Additionally, increased ammonium and decreased dissolved oxygen contents are often observed beneath benthic barriers (Carter et al. 1994; Ussery et al. 1997). These water chemistry considerations may partially explain decreases in macroinvertebrate populations (Engel 1983; Payne et al. 1993) and ammonium content is likely to increase with sediment organic content (Eakin 1992). Toxic methane gas has also been found to accumulate beneath benthic barriers (Gunnison and Barko 1992).

There may be some positive ecological aspects of benthic barriers. Barriers may reduce turbidity and nutrient release from sediments (Engel 1983). They may also provide channels that improve ease of fish foraging when other aquatic plant cover is present near the managed area. Fish may feed on the benthic organisms colonizing any sediment accumulating on top of the barrier (Payne et al. 1993). Payne et al. (1993) also suggest that, despite negative impacts in the managed area, the overall impact of benthic barriers is negligible since they typically are only utilized in small areas of the littoral zone. However, further research is needed on the effects of benthic barriers on fish and wildlife populations and their ability to rebound following barrier removal (Eichler et al. 1995).

S.3.4.4. Dredging

Dredging is a method that involves the removal of top layers of sediment and associated rooted plants, sediment-dwelling organisms, and sediment-bound nutrients. This approach is "non-selective" (USACE 2012), meaning that it offers limited control over what material is removed. In addition to being employed as an APM technique, dredging is often used to manage water flow, provide navigation channels, and reduce the chance of flooding (USACE 2012). Due to the expense of this method, APM via dredging is often an auxiliary effect of dredging performed for other purposes (Gettys et al. 2014). However, reduced sediment nutrient load and decreased light penetration due to greater depth post-dredging may result in multi-season reductions in plant biomass and density (Gettys et al. 2014).

Several studies discuss the utility of dredging for APM. Dredging may be effective in controlling species that propagate by rhizomes, by removing the rhizomes from the sediment before they have a chance to grow (Dall Armellina et al. 1996b). Additionally, invasive phragmites has been controlled in areas where dredging increases water depth to ≥ 5-6 feet; though movement of the equipment used in dredging activities has been implicated in expanding the range of invasive phragmites (Gettys et al. 2014). In streams, dredging resulted in a significant reduction in plant biomass (≥ 90%). However, recovery of plant populations reflected the timing of management actions relative to flowering: removal prior to flowering allowed for plant population recovery within the same growing season, while removal after flowering meant populations did not rebound until the next spring (Kaenel and Uehlinger 1999). Sediment testing for chemical residue levels high enough to be considered hazardous waste (from historically used sodium arsenite, copper, chromium, and other inorganic compounds) should be conducted before dredging, to avoid stirring of toxic material into the water column. The department routinely requires sediment analysis before dredging begins and destination approval of spoils to prevent impacts from sediment leachate outside of the disposal area. Planning and testing can be an extensive component to a dredging project.

<u>Ecological effects of Dredging</u>

Repeated dredging may result in plant communities consisting of populations of fast-growing species that are capable of rebounding quickly (Sand-Jensen et al. 2000). In experimental studies, faster growing invasive plant species with a higher tolerance for disturbance were able to better recover from simulated dredging than slower growing native plant species, suggesting that post-dredging plant communities may be comprised of undesirable invasives (Stiers et al. 2011).

Macroinvertebrate biomass has been shown to decrease up to 65% following dredging, particularly among species which use plants as habitat. Species that live deeper in sediments, or those that are highly mobile, were less affected. As macroinvertebrates are valuable components of aquatic ecosystems, it is recommended that plant removal activities consider impacts on macroinvertebrates (Kaenel and Uehlinger 1999). Dredging can also result in declines to native mussel populations (Aldridge 2000).

Impacts to fish and water quality parameters have also been observed. Dredging to remove aquatic plants significantly increased both dissolved oxygen levels and the number of fish species found inhabiting farm ponds (Mitsuo et al. 2014). This increase in fish abundance may have been due to extremely high pre-dredging density of aquatic plants, which can negatively influence fish foraging success. In another study, aquatic plant removal decreased the amplitude of daily oxygen fluctuations in streams. However, post-dredging changes in metabolism were short-lived, suggesting that algae may have taken over primary productivity (Kaenel et al. 2000). Finally, several studies have also documented or suggested a reduction in sediment phosphorous levels after dredging, which may in turn reduce nutrient availability for aquatic plant growth (Van der Does et al. 1992; Kleeberg and Kohl 1999; Meijer et al. 1999; Søndergaard et al. 2001; Zuccarini et al. 2011). However, consideration must be given to factors affecting whether goals are obtainable via dredging (e.g., internal or external phosphorus inputs, water retention time, sediment characteristics, etc.).

S.3.4.5. Drawdown

Water-level drawdown is another approach for aquatic plant control as well as aquatic plant restoration. Exposure of aquatic plant vegetation, seeds, and other reproductive structures may reduce plant abundance by freezing, drying, or consolidation of sediments. This management technique is not effective for control of all aquatic plant species. Due to potential ecological impacts, it is necessary to consider other factors such as: waterfowl habitat, fisheries enhancement, release of nutrients and solids downstream, and refill and sediment consolidation potential. Often drawdowns for aquatic plant control and/or restoration can be coordinated to time with dam repair or repair of shoreline structures. A review by Cooke (1980), suggests drawdown can provide at least short-term aquatic plant control (1-2 years) when the target species is vulnerable to drawdown and where sediment can be dewatered under rigorous heat or cold for 1-2 months. Costs can be relatively low when a structure for manipulating water level is in place (otherwise high capacity pumps must be used). Conversely, costs can be high to reimburse an owner for lost power generation if the water control structure produces hydro-electric power. The aesthetic and recreational value of a waterbody may be reduced during a drawdown, as large areas of sediment are exposed prior to revegetation. Bathymetry is also important to consider, as small decreases in water level may lead to drop-offs if a basin does not have a gradual slope (Cooke 1980). The downcutting of the stream to form a new channel can also release high amounts of solids and organic matter that can impair water quality downstream. For example, in July 2005, the Waupaca Millpond, Waupaca Co. had to conduct an emergency drawdown that resulted in the river downcutting a new channel. High suspended solid concentrations and BOD resulted in decreased water clarity, sedimentation and depressed dissolved oxygen levels. A similar case occurred in 2015 with the Amherst Mill Pond, Portage Co. during a drawdown at a rate of six inches per day (Scott Provost [WDNR], *personal communication*).

Because extreme heat or cold provide optimal conditions for aquatic plant control, drawdowns are typically conducted in the summer or winter. Because of Wisconsin’s cold winters, winter drawdown is likely to have several advantages when used for aquatic plant management, including avoiding many conflicts with recreational use, potential for cyanobacterial blooms, and terrestrial and emergent plant growth in sediments exposed by reduced water levels (ter Heerdt and Drost 1994; Bakker and Hilt 2016).

A synthesis of the abiotic and biotic responses to annual and novel winter water level drawdowns in littoral zones of lakes and reservoirs is summarized by Carmignani and Roy 2017. Climatic conditions also determine the capacity of a waterbody to support drawdown (Coops et al. 2003). Resources managers pursuing drawdown must carefully calculate the waterbody's water budget and the potential for increased cyanobacterial blooms in the future may reduce the number of suitable waterbodies (Callieri et al. 2014). Additionally, mild winters and groundwater seepage in some waterbodies may prevent dewatering, leading to reduced aquatic plant control (Cooke 1980). Complete freezing of sediment is more likely to control aquatic plants. Sediment exposure during warmer temperatures (>5° C) can also result in the additional benefit of oxidizing and compacting organic sediments (Scott Provost and Ted Johnson [DNR], *personal communication*). When drawdowns are conducted to improve migratory bird habitat, summer drawdowns prove to be more beneficial for species of shorebirds, as mudflats and shallow water are exposed to promote the production of and accessibility to invertebrates during late summer months that coincide with southward migration (Herwig and Gelvin-Innvaer 2015). Drawdowns conducted during mid-late summer can result in conditions that are favorable for cattails (*Typha* spp.) germination and expansion. However, cattails can be controlled if certain stressors are implemented in conjunction with a drawdown, such as cutting, burning or herbicide treatment during the peak of the growing season. The ideal situation is to cut cattail during a drawdown and flood over cut leaves when water is raised. However, this option is not always feasible due to soil conditions and equipment limitations.

<u>Ecological Impacts of Water-level Drawdown</u>

Artificial manipulation of water level is a major disturbance which can affect many ecological aspects of a waterbody. Because drawdown provides species-selective aquatic plant control, it can alter aquatic plant community composition and relative abundance and distribution of species (Boschilia et al. 2012; Keddy 2000). Sometimes this is the intent of the drawdown, which creates plant community characteristics that are desired for wildlife or fish habitat. Consecutive annual drawdowns may prevent the re-establishment of native aquatic plants or lead to reduced control of aquatic plant abundance as drawdown-tolerant species begin to dominate the community (Nichols 1975). Sediment exposure can also lead to colonization of emergent vegetation in the drawdown zone. In one study, four years of consecutive marsh drawdown led to dominance of invasive phragmites (*Phragmites australis subsp. australis*; ter Heerdt and Drost 1994). However, when drawdowns are conducted properly, it can provide a favorable response to native emergent plants for providing food and cover for migrating waterfowl in the fall. Population increases in emergent plant species such as bulrush (*Schoenoplectus* spp.), bur-reeds (*Sparganium* spp.), and wild rice (*Zizania palustris*) is often a goal of drawdowns, which provides a great food source for fish and wildlife, and provides important spawning and nesting habitat. Full or partial drawdowns that are conducted after wild rice production in the fall tend to favor early successional emergent germination such as wild rice and bulrush the following spring. Spring drawdowns are also possible for producing wild rice but must be done during a tight window following ice-out and slowly raised prior to the wild rice floating leaf stage.

Drawdown can also have various effects on ecosystem fauna. Drawdowns can influence the mortality, movement and behavior of native freshwater mussels (Newton et al. 2014). Although mussels can move with lowering water levels, they can be stranded and die if they are unable to move fast enough or get trapped behind logs or other obstacles (WDNR et al. 2006). Some mussels will burrow down into the mud or sand to find water but can desiccate if the water levels continue to lower (Watters et al. 2001). Maintaining a slow drawdown rate can allow mussels to respond and stranded individuals can be relocated to deeper water during the drawdown period to reduce mussel death (WDNR et al. 2006). Macroinvertebrate communities may experience reduced species diversity and abundance from changes to their environment due to drawdown and loss of habitat provided by aquatic plants (Wilcox and Meeker 1992; McEwen and Butler 2008). These effects may be reduced by considering benthic invertebrate phenology in determining optimal timing for drawdown release. Adequate moisture is required to support the emergence of many macroinvertebrate species and complete drawdown may also result in hardening of sediments which can trap some species (Coops et al. 2003). Reduced macroinvertebrate availability can have negative effects on waterfowl and game fish species which rely on macroinvertebrate food sources (Wilcox and Meeker 1992). Depending on the time of year, drawdown may also lead to decreased reproductive success of some waterfowl through nest loss, including common loon (*Gavia immer*) and red-necked grebe (*Podiceps grisegena*; Reiser 1998). However, drawdown may lead to increased production of annual plants and seed production, thereby increasing food availability for brooding and migrating waterfowl. Semi-aquatic mammals such as muskrats and beavers may also be adversely affected by water level drawdown (Smith and Peterson 1988, 1991). DNR Wildlife Management staff follow guidance to ensure drawdowns are timed with the seasons or temperature to minimize negative impacts to wildlife. Negative impacts to reptiles are possible during the spring if water is raised following a drawdown, as nests may be flooded. In the fall, negative impacts to reptiles and amphibians are possible if water is lowered when species are attempting to settle into sediments for hibernation. The impact may be reduced dissolved oxygen if they are below the water or freezing if the water is dropped below the point of hibernation (Herwig and Smith 2016a, 2016b). Surveying and relocation of stranded organisms may help to mitigate some of these impacts. In Wisconsin there are general provisions for conducting drawdowns for APM that are designed to mitigate or even eliminate potential negative impacts.

Water chemistry can also be affected by water level fluctuation. Beard (1973) describes a substantial algal bloom occurring the summer following a winter drawdown which provided successful aquatic plant control. Other studies reported reduced dissolved oxygen, severe cyanobacterial blooms with summer drawdown, or increased nutrient concentrations and reduced water clarity during summer drawdown for urban water supply (Cooke 1980; Geraldes and Boavida 2005; Bakker and Hilt 2016). Water clarity and trophic state may be improved when drawdown level is similar to a waterbody's natural water level regime (Christensen and Maki 2015).

<u>Species Susceptibility to Water-level Drawdown</u>

Not all plant species are susceptible to management by water level drawdown and some dry- or cold-tolerant species may benefit from it (Cooke 1980). Generally, plants and charophytes which reproduce primarily by seed benefit from drawdowns while those that reproduce vegetatively tend to be more negatively affected. Marsh vegetation can be dependent on water level fluctuation (Keddy and Reznicek 1986). Cooke (1980) provides a summary table of drawdown responses for 63 aquatic plant species. Watershield (Brasenia schreberi), fern pondweed (*Potamogeton robbinsii*), pond-lilies (*Nuphar* spp.) and watermilfoils (*Myriophyllum* spp.) tend to be controlled by drawdown. Increases in abundance associated with drawdown have often been seen for duckweed (*Lemna minor*), rice cutgrass (*Leersia oryzoides*) and slender naiad (*Najas flexilis*; Cooke 1980). One study showed drawdown reduced Eurasian watermilfoil (*Myriophyllum spicatum*) at shallow depths while another cautioned that Eurasian watermilfoil vegetative fragments may be able to grow even after complete desiccation (Siver et al. 1986; Evans et al. 2011). Similarly, a tank-simulated drawdown experiment suggested short-term summer drawdown may be effective in controlling monoecious hydrilla (*Hydrilla verticillata*; Poovey and Kay 1998). However, other studies have shown hydrilla fragments to be resistant to drying following drawdown (Doyle and Smart 2001; Silveira et al. 2009). A study on Brazilian waterweed (*Egeria densa*) showed that stems were no longer viable after 22 days of exposure due to drawdown (Dugdale et al. 2012).

Two examples of recent drawdowns in Wisconsin that were evaluated for their efficacy in controlling invasive aquatic plants occurred in Lac Sault Dore and Musser Lake, both in Price County, which were conducted in 2010 and 2013, respectively. Dam maintenance was the initial reason for these drawdowns, with the anticipated control of nuisance causing aquatic invasive species as a secondary benefit. Aquatic plant surveys showed that the drawdown in Lac Sault Dore resulted in a 99% relative reduction in the littoral cover of Eurasian watermilfoil when comparing pre- vs. post-drawdown frequencies. Native plant cover expanded following the drawdown and Eurasian watermilfoil cover has continued to remain low (82% relative reduction compared to pre-drawdown) as of 2017 (Onterra 2013). Lake-wide cover of curly-leaf pondweed in Musser Lake decreased following drawdown (63% relative reduction compared to pre-drawdown), and turion viability was also reduced. Reductions in native plant populations were observed, though population recovery could be seen in the second year following the drawdown (Onterra 2016). These examples of water-level drawdowns in Wisconsin show that they can be valuable approaches for aquatic invasive species control in some waterbodies. Water level reduction must be conducted such that a sufficient proportion of the area occupied by the target species is exposed. Numerous other single season winter drawdowns monitored in central Wisconsin by department staff show similar results (Scott Provost [DNR], *personal communication*). Careful timing and proper duration is needed to maximize control of target species and growth of favorable species.

S.3.5.Biological Control

Biological control refers to any method involving the use of one organism to control another. This method can be applied to both invasive and native plant populations, since all organisms experience growth limitation through various mechanisms (e.g., competition, parasitism, disease, predation) in their native communities. As such, when control of aquatic plants is desired it is possible that a growth limiting organism, such as a predator, exists and is suitable for this purpose.

Care must be taken to ensure that the chosen biological control method will effectively limit the target population and will not cause unintended negative effects on the ecosystem. The world is full of examples of biological control attempts gone wrong: for example, Asian lady beetles (*Harmonia axyridis*) have been introduced to control agricultural aphid pests. While the beetles have been successful in controlling aphid populations in some areas, they can also outcompete native lady beetles and be a nuisance to humans by amassing on buildings (Koch 2003). Additionally, a method of control that works in some Wisconsin lakes may not work in other parts of the state where differing water chemistry and/or biological communities may affect the success of the organism. The department recognizes the variation in control efficacy and well as potential unintentional effects of some organisms and is very cautious in allowing their use for control of aquatic plants.

Purple loosestrife beetles

The use of herbivorous insects to reduce populations of aquatic plants is another method of biocontrol. Several beetle species native to Eurasia (*Galerucella calmariensis*, *G. pusilla*, *Hylobius transversovittatus*, and *Nanophyes marmoratus*) have been well-studied and intentionally released in North America for their ability to suppress populations of the invasive wetland plant, purple loosestrife (*Lythrum salicaria*). These beetles only feed on loosestrife plants and therefore are not a threat to other wetland plant species (Kok et al. 1992; Blossey et al. 1994a, 1994b; Blossey and Schroeder 1995). The department implements a purple loosestrife biocontrol program, in which citizens rear and release beetles on purple loosestrife stands to reduce the plants' ability to overtake wetlands, lakeshores, and other riparian areas.

Beetle biocontrol can provide successful long-term control of purple loosestrife. The beetles feed on purple loosestrife foliage which in turn can reduce seed production (Katovich et al. 2001). This approach typically does not eradicate purple loosestrife but stresses loosestrife populations such that other plants are able to compete and coexist with them (Katovich et al. 1999). Depending on the composition of the plant community invaded by purple loosestrife and the presence of other non-native invasive species, further restoration efforts may be needed following biocontrol efforts to support the regrowth of beneficial native plants (McAvoy et al. 2016).

Several factors have been identified that may influence the efficacy of beetle biocontrol of purple loosestrife. Purple loosestrife beetles have for the most part been shown to be capable of successfully surviving and establishing in a variety of locations (Hight et al. 1995; McAvoy et al. 2002; Landis et al. 2003). The different species have different preferred temperatures for feeding and reproduction (McAvoy and Kok 1999; McAvoy and Kok 2004). In addition, one study suggests that the number of beetles introduced does not necessarily correlate with greater beetle colonization (Yeates et al. 2012). Disturbance, such as flooding and predation by other animals on the beetles, can also reduce desired effects on loosestrife populations (Nechols et al. 1996; Dech and Nosko 2002; Denoth and Myers 2005). Finally, one study suggests that the use of triclopyr amine for purple loosestrife control may be compatible with beetle biocontrol, although there may be negative effects on beetle egg-batch size or indirect effects if the beetle's food source is too greatly depleted (Lindgren et al. 1998). Some mosquito larvicides may harm purple loosestrife beetles (Lowe and Hershberger 2004).

Milfoil weevils

Similar to the use of beetles for biological control of purple loosestrife, the use of milfoil weevils (*Euhrychiopsis lecontei*) has been investigated in North America to control populations of non-native Eurasian and hybrid watermilfoils (*Myriophyllum spicatum* x *sibiricum*). This weevil species is native to North America and is often naturally present in waterbodies that contain native watermilfoils, such as northern watermilfoil (*M. sibiricum*). The weevils have the potential to damage Eurasian watermilfoil (*M. spicatum*) by feeding on stems and leaves and/or burrowing into stems. Weevils may reduce milfoil plant biomass, inhibit growth, and compromise buoyancy (Creed and Sheldon 1993; Creed and Sheldon 1995; Havel et al. 2017a). Damage caused to the milfoil tissue may then indirectly increase susceptibility to pathogens (Sheldon and Creed 1995).

In experiments, weevils have been shown to negatively impact Eurasian watermilfoil populations to varying degrees. Experiments by Creed and Sheldon (1994) found that plant weight was negatively affected when weevils were at densities of 1 and 2 larvae/tank, and Eurasian watermilfoil in untreated control tanks added more root biomass than those in tanks with weevils, suggesting that weevil larvae may interfere with the plant's ability to move nutrients. Similarly, experiments by Newman et al. (1996) found that weevils at densities of 6, 12, and 24 adults/tank caused significant decreases in Eurasian watermilfoil stem and root biomass, and that higher weevil densities generally produced more damage.

In natural communities, effects of weevils have been mixed, likely because waterbody characteristics may play a role in determining weevil effects on Eurasian watermilfoil populations in natural lakes. In a 56 ha (138 acre) pond in Vermont, weevil density was negatively associated with Eurasian watermilfoil biomass and distribution; Eurasian watermilfoil beds were reduced from 2.5 (6.2 acres) to 1 ha (2.5 acres) in one year, and biomass decreased by 4 to 30 times (Creed and Sheldon 1995). A survey of Wisconsin waterbodies conducted by Jester et al. (2000) revealed that most lakes containing Eurasian watermilfoil also contained weevils. Weevil abundance varied from functionally non-detectable to 2.5 weevils/stem and was positively associated with the presence of large, shallow Eurasian watermilfoil beds (compared to deep, completely submerged beds). There was no relationship between natural weevil abundance and Eurasian watermilfoil density between lakes. However, when the authors augmented natural weevil populations in plots in an attempt to achieve target densities of 1, 2, or 4/stem, they found that augmentation was associated with significant decreases in Eurasian watermilfoil biomass, stem density and length, and tips/stem (Jester et al. 2000). However, another more recent study conducted in several northern Wisconsin lakes found no effect of weevil stocking on Eurasian watermilfoil or native plant biomass (Havel et al. 2017a).

There are several factors to consider when determining whether weevils are an appropriate method of biocontrol. First, previous research has suggested that densities of at least 1.5 weevils per stem are required for control (Newman and Biesboer 2000). Adequate densities may not be achievable due to factors including natural population fluctuations, the amount of available milfoil biomass within a waterbody, the presence of insectivorous predators, such as bluegills (*Lepomis macrochirus*), and the availability of nearshore overwintering habitat (Thorstenson et al. 2013; Havel et al. 2017a). In addition, weevils fed and reproduce on native milfoil species and biocontrol efforts could potentially impact these species, although experiments conducted by Sheldon and Creed (2003) found that native milfoil weevil density was lower and weevils caused less damage than when they were found on Eurasian watermilfoil. Adult weevils spend their winters on land, so available habitat for adults must be present for a waterbody to sustain weevil populations (Reeves and Lorch 2011; Newman et al. 2001). Additionally, one study found that lakes with no Eurasian watermilfoil (despite the presence of other milfoil species) and lakes that had a recent history of herbicide treatment had lower weevil densities than similar, untreated lakes or lakes with Eurasian watermilfoil (Havel et al. 2017b).

Grass carp – not allowed in Wisconsin

The use of grass carp (*Ctenopharyngodon idella*) to control aquatic plants is not allowed in Wisconsin; they are a prohibited invasive species under ch. NR 40, Wis. Admin. Code, which makes it illegal to possess, transport, transfer, or introduce grass carp in Wisconsin.

Sterile (also known as triploid) grass carp have been used to control populations of aquatic plants with varying success (Pípalová 2002; Hanlon et al. 2000). Whether this method is effective depends on several factors. For instance, each individual fish must be tested to ensure sterility before stocking, which can be a time- and resource-consuming process. Since the sterile fish do not reproduce, it can be difficult to achieve the desired density in a given waterbody. In addition, grass carp, like many fish species, have dietary preferences for different plant species which must be considered (Pine and Anderson 1991). Further information summarizing the effects of stocking triploid grass carp can be found in Pípalová (2006), Dibble and Kovalenko (2009), and Bain (1993).